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Economic instruments and waste policies in the Netherlands

Inventory and options for extended use

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Abstract

In recent years, the interest in the use of economic instruments in environmental policy has been growing, reflecting increasing awareness of their potential cost-effectiveness as well as the need to diversify the ‘policy toolbox’. Waste policy is no exception to this tendency.

The present study explores the opportunities for extended use of economic instruments for waste policy in the Netherlands, focusing on waste from households and the trade, services and government sector. Five economic instruments are specifically addressed:

1. Waste taxes
2. Waste collection charges
3. Taxes on raw materials and products
4. Deposit-refund schemes
5. Subsidies and fiscal incentives

For each of these economic instruments, the study describes existing experiences (both in the Netherlands and abroad) and analyses the impact and feasibility of a variety of possible new applications. These case studies, which primarily serve as illustrations of the mechanisms and possible impacts, were selected so as to represent a wide range of instrument types, levels of intervention, positions in the value chain, and types of waste. Depending on the application area and data availability, the analysis is done by means of quantitative approaches (using general-equilibrium modelling and statistical analysis) or more qualitative approaches in which expert judgements are used.

Waste taxes

Several countries apply taxes on final waste disposal. In the Netherlands, this tax is limited to landfilling, and its rate is the highest in the EU. In some other countries there is a tax on incineration as well. The available evidence on the effectiveness of waste taxes is limited. The Dutch landfill tax has made landfilling more expensive than incineration, resulting in more recycling by companies in the service sector. In some countries landfill taxes have clearly led to a reduction in the landfilling of construction and demolition waste (CDW).

The following waste tax options were analysed in this study:

- *Applying the standard landfill tax rate (instead of the current reduced rate) to CDW with a gypsum content of more than 1%.* This measure would enhance the separation of gypsum from other CDW. However, in order to stimulate gypsum recycling additional measures will be needed, as there may otherwise still be cheaper destinations for the separated gypsum (such as export for mine filling). Without such complementary policies, the environmental benefits of this option are doubtful.
- *Introducing a tax on incineration and/or increasing the rate for landfilling.* Under current conditions, these options would have only marginal impacts on the amounts of waste landfilled, incinerated and recycled. An important reason is that most

households do not ‘feel’ the incentive of the waste tax due to a lack of differential and variable rate (DVR) charging for waste collection (as discussed below). Other reasons for the small impact of variations in waste tax include the limited incineration capacity and the high costs of (additional) recycling. The latter also implies that an incineration tax may lead to more export of waste rather than more recycling. Furthermore, differentiation of the incineration tax rate by energy efficiency of the incineration plant will only be effective if there are no capacity restrictions, which includes restrictions like high entry costs and public resistance to the construction of new incineration plants.

Waste collection charges

The use of differential and variable rates (DVR) in waste collection charging can be seen as a necessary complement to waste taxes. DVR ensure that the price incentive of waste taxes are transmitted to households and thus contributes to waste reduction and recycling. International empirical evidence confirms the effectiveness of DVR schemes. Weight-based and pay-per-bag schemes appear to perform equally well, but the latter are less costly. An essential condition for DVR to be effective is the presence of a comprehensive system for the collection of segregated materials for recycling. Although the effects on illegal waste disposal are difficult to estimate, the net social benefits of DVR are likely to be positive in most cases.

In the Netherlands, currently some 20% of the population is charged at DVR. In other countries (such as Belgium) this percentage is much higher. South Korea successfully applies a mandatory DVR system nation-wide since 1995.

In the present study, various scenarios for extended use of DVR in the Netherlands have been analysed. In each scenario the penetration of DVR in highly urbanized areas is assumed to be lower than in less urbanized areas. The model analyses take into account differences between types of households and dwellings. In the scenario with the highest share of inhabitants under DVR (56%), the total amount of waste will decrease by 3%. The total amount of residual solid waste will decrease by 11.6% and the organic waste (which is charged at a lower rate) by 5%. The recycling of glass and paper will increase by 11% and 21%, respectively.

Based on earlier studies, such as CE 2008, an assessment was made on the environmental impacts of the economic instruments. The environmental impacts of increased DVR charging are mixed. Several environmental themes show improvements, but primary energy use and greenhouse gas emissions increase, due to the reduced supply of waste for energy production, which presumably has to be replaced by primary (fossil) energy.

Taxes on (raw) materials and products

From a theoretical point of view, taxes on (raw) materials and products are not the most efficient instruments for waste prevention and reduction, but in practice they may be useful tools to stimulate recycling, reduce material intensity, and promote the use of environmentally preferable substitutes. Many countries, including the Netherlands, apply taxes or charges to packaging (materials). Taxes and charges on aggregates (sand, gravel etc.), disposable products, batteries and electr(on)ic appliances are also quite common.

Besides their incentive function (increasing the price of the product or material) these levies often also serve as a source of revenue for the treatment of the associated waste.

The following options for extended application of this instrument in the Netherlands have been analysed:

- *A tax on sand and gravel extraction.* International experience with this type of tax shows a limited effectiveness in terms of overall demand reduction and higher recycling rates. Moreover, it may lead to substitutions (e.g. to crushed rock) that are not necessarily environmentally sound. To avoid substitution by imports, the tax should preferably be applied to domestic and imported materials alike.
- *Taxes on paper and cardboard, aluminum and packaging.* Model simulations have been carried out with tax rates that lead to a doubling of the price of the virgin material (which is substantially higher than the current packaging tax rates). The main impacts of this measure will be an increase in product prices and an increase in the import of recycled materials. The impact on waste separation for recycling is very limited: the highest increase is 3.5% (in the repair sector). Again, households that are not charged at DVR will not show any response in terms of waste separation. It is concluded that this type of tax would be far more effective if it would be accompanied by measures, such as DVR or recycling subsidies, to increase waste separation by the service sector and households.

Deposit-refund schemes

Deposit-refund schemes (DRS) can be efficient policy instruments to encourage reuse and recycling. DRS are widely applied in the area of beverage packaging, mainly on a voluntary, but sometimes also on a mandatory basis. Generally, DRS systems lead to high return rates and a reduction of littering. On the other hand, the handling and administration costs can be substantial.

Using a quantitative model, the impact of introducing mandatory DRS has been analysed for the following product categories:

- *Small electric appliances (white goods):* a deposit-refund rate of €5 (resp. €15) per appliance would lead to an increase in the number of recycled appliances. The recycling rate (recycled appliances as a percentage of total amount of appliances disposed of) would increase from the current 60.7% to between 64.7 and 76.4%.
- *Batteries:* a deposit-refund rate of €5 (resp. €20) per kg (i.e. €0.12 to €0.52 per battery, ranging from some 5 to 50% of its price) would lead to an increase in the total amount of batteries separately collected and recycled. The recycling rate would increase from the current 86.9% to between 87.2 and 89.2%.

Obviously, the performance of a DRS in terms of additional recycling is stronger in cases where current recycling rates are relatively low. Moreover, the pre-existence of an infrastructure for separate collection would make small white goods an interesting candidate for this instrument.

Subsidies and fiscal incentives

Subsidies and other kinds of financial support can be useful policy instruments in situations where market imperfections (such as high transaction and monitoring costs, or distorted product markets) preclude the use of ‘first best’ tools such as waste disposal fees. Presently, the Netherlands as well as other countries apply a wide variety of subsidies in their waste policies.

In the present study, the following options for ‘positive financial incentives’ in waste policy were analysed:

- *Subsidies for food banks:* Apart from social motives, subsidies could be justified as contributions to the prevention of food waste. However, the amounts involved are likely to be insignificant when compared to the total amount of food waste.
- *Food waste reduction as an award criterion in public tenders:* By rewarding caterers who succeed in reducing food waste, public authorities can create a significant incentive in this waste chain, given their share in market demand.
- *Reduced VAT rates for second-hand goods:* Due to a lack of data, little can be said about the potential effectiveness of this measure. In any case, it would require a change in EU rules. Moreover, the size of the associated price reduction is limited (maximum 13%) and the environmental merits of prolonging the life of certain products are equivocal. A reduced VAT rate for ‘intrinsically waste-extensive’ products (e.g. reusable nappies) could be more effective.

Conclusions and recommendations

By combining a literature review, an inventory of foreign experiences and economic analysis, this study shows that economic instruments have an important role to play in waste management. The case studies presented in this report mainly illustrate the underlying mechanisms and possible impacts of economic instruments in waste policy, applied to a selective number of product-waste chains. Any generalisations should therefore be treated with caution.

Despite this reservation, it can be concluded that there is scope for a wider and more intense use of economic instruments in waste policy in the Netherlands. In theory, economic instruments generate strong incentives to make people behave in a more environmentally friendly way, and to move waste streams to higher levels in the waste hierarchy. Dutch waste policy is currently characterised by a mixture of direct regulation, economic instruments and other instruments (such as voluntary agreements and information provision). This study demonstrates that there is a clear potential for expanding application of economic instruments in the Netherlands.

However, the potential of economic instruments is limited by a number of ‘real world’ conditions. First, price incentives are often not reaching the actors who have to change their behaviour. This is partly caused by inefficient design of economic instruments, for instance, in the case where price signals intervene too early in the product chain to have an effect on the waste stage. Otherwise, price incentives may be distorted due to the absence of a mechanism to transfer incentives to households and the service sector. This is the case for the waste tax in the Netherlands which is not always effectively passed on

to the waste generators. Second, in many cases recycling is still more costly than the conventional way of managing waste. Therefore, making recycling more attractive may sometimes require substantial political support to release additional public funds. This support is not always there. Finally, economic instruments never operate in a closed system. Even at a local scale, leakage of effects may occur, for instance in the case of the increase of illegal dumping as a result of the DVR schemes. At a national and international scale, such leakage effect also may occur (e.g. waste export).

Policy makers should take into account these potential barriers in designing economic instruments. For example, decision makers should take into account the lesson that the instruments should intervene as close to the subject at stake as possible to prevent distortions avoiding the instrument to be effective. Also, policy makers should explicitly map out the existing market imperfections before implementing new economic instruments. In fact, it may sometimes be more effective to change the conditions in which waste policies are operating, rather than add more policies. Nevertheless, it will be inevitable for policy makers to sometimes accept second-best choices, while trying to limit the negative consequences as much as possible by explicitly addressing negative side-effects. Being part of the EU, the freedom of the Dutch government to implement new economic policies is limited.

One way of following a rational and realistic approach in expanding the influence of economic instruments in waste policies in the Netherlands is by explicitly addressing their potential effectiveness and feasibility as well as the conditions that need to be fulfilled and the potential obstacles and limitations. For the instrument categories that have been considered in this study, the main issues can be summarized as follows.

Waste taxes

An increase in the tax on landfilling and the introduction of a non-zero tax rate on incineration will only be effective if simultaneous measures are taken which transfer the incentives of the tax on to the producers of waste (i.e. households and the service sector). This can be achieved, for example, by introducing unit-based pricing (DVR; see below) and stimulating separate collection of waste streams. Moreover, alternative treatment capacity should be sufficient to allow for switching to the most desirable waste treatment option. There should also be sufficient confidence that the lowest-cost alternatives (including those abroad) are indeed the environmentally preferable ones. Further research is needed to explore to what extent these conditions are met or can be met in the near future.

Waste collection charges

DVR has proven to be (under certain conditions) a cost-effective instrument for waste reduction and more recycling. Moreover, DVR can itself be seen as a necessary condition for the successful implementation of other instruments, such as waste taxes (see above). It is therefore recommended to stimulate municipalities to adopt DVR schemes, accompanied by the provision of adequate facilities for waste separation by households. Before introducing DVR in 'very strongly urbanized' municipalities (i.e. the 12 largest cities in the Netherlands) a pilot experiment in one of them might be useful to test the behavioural response in this category.

Taxes on raw materials and products

The introduction of new taxes on specific raw materials and products, and the increase in the tax rate for existing ones (such as the packaging tax) can in principle contribute to higher recycling rates and a lower resource intensity in general. However, the presence of other incentives to increase the supply of recycled materials is essential for the effectiveness of such taxes. Moreover, (very) high tax rates will be needed to achieve any significant impact, and such high rates may cause unwanted side effects (evasion; substitution by untaxed, but equally undesirable alternatives). Revenue raising rather than waste reduction might therefore be the main consideration justifying the use of such taxes (at relatively low rates).

Deposit refund systems

Deposit refund systems (DRS) have a strong potential to achieve high collection and recycling rates. Their application therefore deserves consideration, especially with respect to (discarded) products for which separate collection rates are currently low, while an adequate infrastructure is already in place or can be created without excessive cost. Small electric appliances may be a case in point. Before introducing (mandatory) DRS, their impact on international trade and interactions with other policies (e.g. in the framework of the WEEE Directive) should be further investigated. The administrative costs and overall environmental impact would also require more detailed analysis.

VAT reduction, subsidies and public procurement

Positive financial incentives for ‘low-waste’ products and services can be justifiable exemptions to the ‘polluter pays principle’ under certain circumstances. They can be given in a number of different ways. Among the examples analysed in this study, the inclusion of waste reduction clauses in public tenders has been identified as a promising candidate. The potential effectiveness and feasibility of subsidies and other positive incentives should be analysed on a case-by-case basis. Their introduction may depend on consent at EU level (fiscal harmonisation; state aid rules).

1. Introduction

Environmental policies generally include a balanced mix of complementary measures including regulatory, economic, educational and informative instruments. Especially economic instruments, such as taxes, unit pricing and deposits, gain popularity among policy makers for different reasons. On the one hand, these instruments are able to contribute to financing waste management activities. On the other hand, they have the capacity of persuading households and producers to strive towards diverting waste from landfills, recycle more waste and optimise the use of resources.

In the last 15 years, the Netherlands has expanded the use of economic instruments in its environmental policies. The waste sector is no exception to this trend. Economic instruments are also more commonly applied in various segments of the production and consumption chain with the aim to reduce the environmental impact of waste. In some respects, the Netherlands is applying these instruments more intensively than other countries. For example, the tax rate for landfilling waste is higher than in any other EU Member State. On the other hand, there are several areas (such as the application of differential charging for waste collection) where other countries are ahead of the Netherlands.

Mid 2008 the Dutch ministry of VROM wanted to gain an insight whether there are still unused opportunities to exploit the capacity of economic instruments to deliver efficient solutions for waste problems in the Netherlands. This resulted in a study carried out by the Institute for Environmental Studies (IVM) and the Agricultural Economic Institute (LEI). The overall aim of the study is to explore the opportunities for extended use of economic instruments for waste policy in the Netherlands. This is achieved by:

1. Making an inventory of the current practice of economic instruments in waste management in the Netherlands and abroad and evaluating the Dutch and international examples to learn more about the efficiency of these applications in waste management. This part of the study is mainly based on literature research;
2. Analyzing the economic and environmental impact and the feasibility of a number of economic instruments for a range of waste materials. These options have been selected so as to cover a variety of instruments and 'points of interference' in the waste chain, mostly focusing on waste from households and the trade, services and government sectors.¹ Various methodologies are applied to fulfil this task, ranging from quantitative approaches using general-equilibrium modelling and statistical analysis to qualitative approaches in which expert judgements are used.

This report is structured as follows. Chapter 2 describes the theoretical and policy background of economic instruments in the waste sector. The next chapters are devoted to the various types of economic instruments: waste taxes (Chapter 3); waste collection

¹ The main function of the selected options is to illustrate the mechanisms at work and the direct and indirect impacts that can be expected from the use of various economic instruments. The exemplary applications are mostly taken from the 'priority waste streams' under VROM's 'chain approach' as announced in the Second Waste Management Plan (LAP2; VROM, 2008).

charges (Chapter 4); taxes on raw materials and products (Chapter 5); deposit-refund schemes (Chapter 6); and subsidies and fiscal incentives (Chapter 7). Each of these chapters describes the current use of the instrument in the Netherlands as well experiences with its use in other countries, and presents the outcome of the analysis of several potential options for extended or improved use of the instrument. Chapter 8 draws conclusions and formulates recommendations.

2. Background

2.1 Introduction

The success of any waste management and recycling policy depends on changes in the behaviour of producers, consumers and waste processors. Authorities can try to bring about these changes by means of various instruments. Unfortunately, there is no generally accepted standardised classification of waste management and recycling policy instruments. However, there is some agreement that three broad categories of environmental instruments can be distinguished (Opschoor and Turner 1994, p.10):

1. *Economic instruments*: instruments that affect the market conditions under which people and firms make their decisions, without directly reducing the decision space available to them.
2. *Direct regulative instruments*: instruments that influence the range of alternatives by means of prohibitions, restrictions or obligations. Certain public investments such as the provision of infrastructure and facilities may broaden the range of available alternatives and therefore also belong to this category. An alternative term for this category is ‘command-and-control’ regulations.
3. *Communicative instruments*: instruments aimed at voluntary adaptations of individual and group behaviour in a more environmentally friendly direction.

This Chapter describes the theoretical and policy background of policy instruments in the waste sector in general, with the aim to better understand the complexities underlying the use of economic instruments. Note that more specific details about the use of economic instruments used in waste policy in the Netherlands and abroad are given in the third and fourth section in each of the Chapters 3 through Chapter 7 and Appendix I in this report.² Moreover, the system boundaries and methodological approach applied in this study are elaborated upon.

2.2 Policy background

Traditionally, environmental policy in most countries has mainly used direct regulative instruments to achieve its objectives and waste policy is no exception to this convention. Examples include bans or restrictions on landfilling, emission standards for incineration plants, and ‘take back obligations’ for discarded goods (producer responsibility). In addition, the public provision of waste disposal facilities (e.g. dustbins, bottle banks) has been an important instrument to avoid littering and support recycling.

In recent years, the interest in and application of economic instruments has been growing. Policy makers have discovered the power of financial incentives to make people behave in a more environmentally friendly way, and to move waste streams to higher levels in the waste hierarchy. Many governments (at various levels) have

² In Addition, Appendix II presents information on trading schemes; a type of instrument that is not addressed in the main text.

introduced taxes and charges on the final disposal of waste and on products generating waste, differential rates for waste collection charges, deposit-refund schemes, and subsidy schemes for waste reduction (see e.g. EEA, 2005b).

The European Commission (e.g. 2005, 2007) has repeatedly stated that it is in favour of using economic instruments more intensively in waste as well as other environmental policies. The Organisation for Economic Co-operation and Development (OECD) is also a strong supporter of applying economic instruments in waste policy (see e.g. OECD, 2004a) and has recommended Member States (including The Netherlands; see OECD, 2003) to extend their use. Dutch waste policy is currently characterised by a mixture of direct regulation, economic instruments and other instruments (such as voluntary agreements and information provision).

Box 2.1 A terminological note on taxes and charges

Taxes and charges are mandatory payments by individuals or companies to a publicly administered account. The term ‘taxes’ is commonly used for those payments for which there is no direct relationship between the payment and the use of the revenues. Generally, tax revenues accrue to the general public budget, although sometimes they may be earmarked for specific purposes (for example, in many Eastern European countries the environmental tax revenues go to Environmental Funds). Charges, on the other hand, are payments for specific services provided by the public body (e.g. the collection of municipal waste), and the revenues from the charge are specifically spent on this purpose. The amounts that private persons and companies have to pay under a charge system are not necessarily proportional to the services provided: for example, households often pay a fixed amount for waste collection (although the use of differential rates is increasing; see Section 4.3).

2.3 Theoretical background

Since many years, economists have developed theoretical arguments for the use of economic instruments in environmental policy. The ‘archetype’ of these instruments is the ‘Pigovian’ tax on pollution, which is set at a level equal to the marginal environmental damage, i.e. the damage caused by an additional unit of pollution. If this damage is unknown, a socially optimal solution can still be achieved by imposing a tax that would induce polluters to reduce their pollution to the socially desired level. Alternatively, the latter level can also be used as the ‘cap’ on total emissions in an emissions trading system, which is in principle equivalent to a tax (though the distributional impact may be different).³

The main reason why pollution taxes (or tradable permits) are from an economic point of view preferable to direct regulation is their greater (static and dynamic) efficiency. Economic instruments will ensure that pollution abatement takes place where this can be done at the lowest cost. Polluters who have only abatement options with costs per unit exceeding the tax rate (or, for that matter, the market price of the tradable permit) will continue to pollute and pay. Moreover, economic instruments provide a lasting incentive for the polluter to search for new abatement options, as each tonne of pollutant emitted carries a price tag.

³ The basic theory of economic instruments is discussed in Baumol and Oates (1988).

It should be noted that the full benefits of using economic instruments will only be reaped if certain conditions are fulfilled. In particular, all actors should be well informed about their options for pollution control and the associated costs, and the market should function smoothly (ensuring, among others, that price incentives are ‘passed on’ along the value chain).

The increasing interest in using economic instruments among policy makers is not merely a matter of discovering that these instruments are (at least in theory and under certain assumptions) superior to direct regulation in terms of static and dynamic efficiency. There is also increasing recognition that the complexity of environmental problems (including waste management) calls for the use of a tailor-made package of instruments, rather than a single, one-size-fits-all solution (see e.g. OECD, 2007c). Hence, besides the efficiency of economic instruments, the growing interest for integrating these instruments in waste policy also results from the general need to broaden the range of policy instruments.

In waste policy, the application of the ‘textbook solution’ of environmental taxation is in particular complicated by the risk of illicit waste dumping (fly-tipping) that might increase if legal disposal options become more expensive. In some cases, the additional monitoring and enforcement costs required to prevent this could outweigh the efficiency gains resulting from the economic instrument.

Another particular feature of waste policy is that it usually serves different policy objectives, such as preventing resource depletion, improving energy efficiency, reducing emissions from landfill sites and incineration plants. When comparing waste policy instruments, it is therefore not sufficient to look at their performance in terms of the primary objective (e.g. x% increase in the recycling rate of a certain waste stream), but one would also like to know their different impacts on various underlying objectives.

2.4 System boundaries

To determine the potential of economic instruments in the Dutch waste policy, an analytical framework has to be designed representing the most relevant conceptual dimensions of waste management. At the same time, the analytical framework should also be able to provide practical advice to Dutch policy makers, who are constrained by former policy decisions and thus are often unable to redesign waste policies from scratch. The balance between theory and practice is demonstrated by addressing several dimensions of the analytical framework.

Scale of intervention

The effectiveness of economic instruments strongly depends on the scale of intervention (see Figure 2.1). On the one hand, for example in the case of unit-based pricing for waste disposal, implementation on a municipal scale is most effective because local authorities are better able to account for local conditions. On the other hand, for example in the case of a deposit-refund scheme, national implementation is more effective given the need for a country wide collection and refund system. One can even imagine that international intervention may be needed in the case of tradable waste and recyclable material flows (for example through harmonisation of import levies) as well as in the case of multinational companies setting up take-back schemes of their used products.

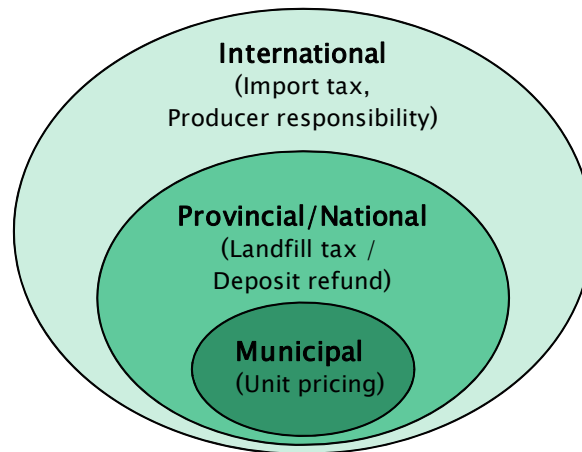


Figure 2.1 Economic instruments intervene at various levels of governance

In this study, intervention levels are selected at the local/municipal scale as well as at the national scale. It should be realised, however, that the Dutch government is not always in the position to determine a waste policy independent of the European Union. In several cases, such as the reduction of VAT for reusable and recyclable waste streams, EU regulations have to be complied with.

Position in the value chain

Traditionally, waste policies are mainly focused at the final phase in the material chains, namely the waste stage. In the Netherlands, this approach resulted in a significant increase in reuse, recycling and energy recovery. However, the additional gains to be made with this traditional approach are limited. Therefore, the new draft waste policy plan of the Netherlands follows a chain approach (VROM, 2008). After all, a chain approach is expected to offer the most cost-efficient improvements towards more sustainable material use in the Netherlands.

The chain approach does not only hold for physical engineering measures across the material chain (e.g. design for recycling, preventing the use of hazardous materials). As shown in Figure 2.2, the chain approach also provides guidance for policy instruments that can intervene at various stages in the chain (i.e. raw material measures, policies focused at the consumption stage, or interventions directly in the waste management stage). The economic instruments depicted in Figure 2.2 are the ones that have been selected as options for analysis in this study. This selection was made in consultation with the contractor of this study (the Ministry of VROM). We made an attempt to select economic instruments across the chain to demonstrate their effectiveness in reducing environmental impact of waste by households and the service sector.

The design of a ‘chain proof’ waste policy implies that interventions in various parts of the chain will not lead to negative effects in the waste segment, and instead improve the waste performance across the chain. Such a chain focus is important because it may very well be the case that interventions early in the chain only show a positive and desirable impact in later parts of the material chain. Economic instruments in Dutch waste policy might be designed in a way that makes them more ‘chain proof’. For example, subsidy programs could add selection criteria in their approval procedure describing the number of stakeholders across the chain participating in the proposal (VROM 2008).

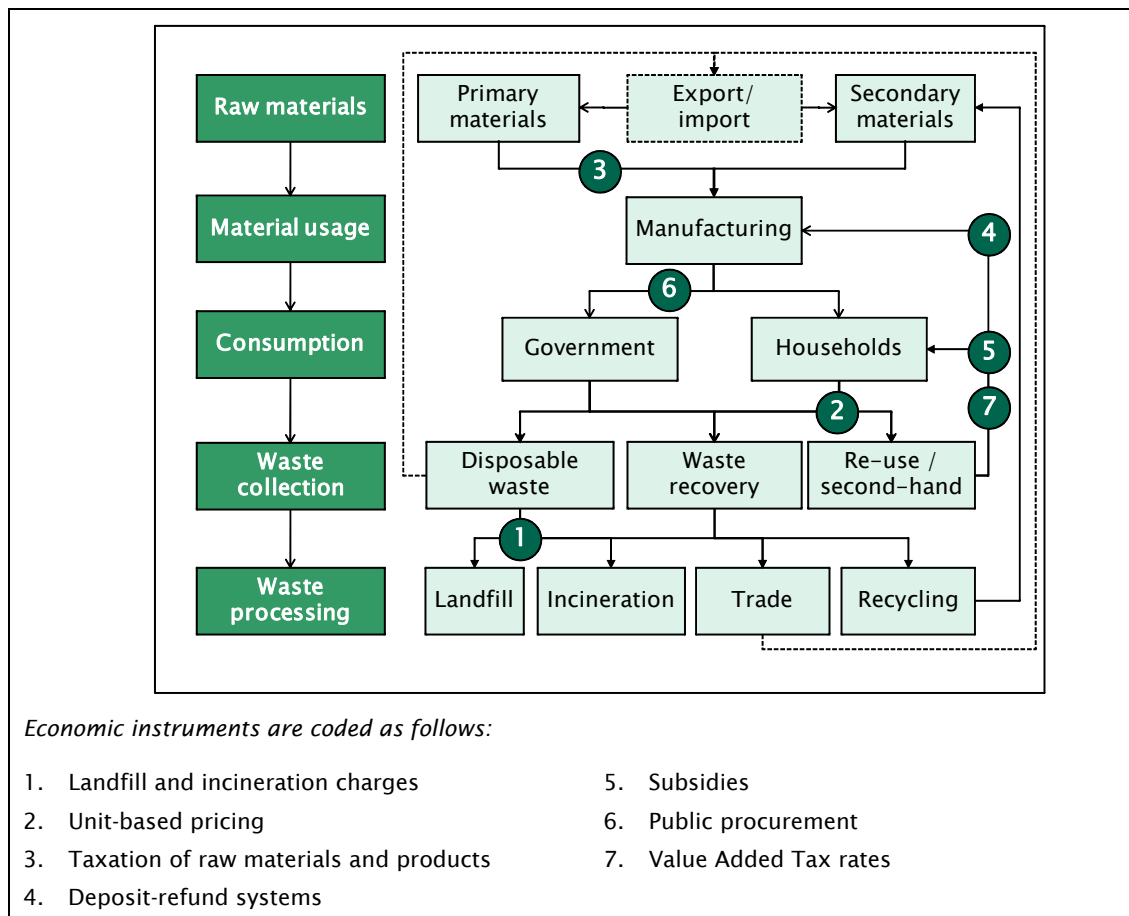


Figure 2.2 Economic instruments intervene at various levels in the value chain

Type of waste

The effectiveness of the economic instruments considered depends greatly on the selection of the type of waste from households and the service sector to which these economics instruments are applied. For example, a deposit refund system may work well for beverage containers (due to its limited life time and short consumption phase) but is probably less effective in limiting disposal of televisions (which have a much longer life time). These implications may result from the physical and practical characteristics of the waste type as well as the existing policy targets set for the particular material. For example, due to their small size, household batteries are generally difficult to address through economic instruments. Most used batteries end up in drawers or are mixed with the residual household waste. For certain types of waste flows in the Netherlands (e.g. organic waste, glass, packaging, e-waste) specific policy goals and requirements have been set, influencing the applicability and added value of economic instruments.

The choice of the various waste materials under consideration is mainly based on the recently identified priority waste streams for the chain approach in Dutch waste policy. These waste flows have been selected as focal waste streams for the coming planning period. The aim of the identification of the priority waste streams is to reduce the environmental pressure of these waste streams with 20% by 2012 (VROM 2008). As mentioned before, the main function of the selected waste materials and economic instruments for this study is to illustrate the mechanisms at work and the direct and

indirect impacts that can be expected from the use of various economic instruments. The combination of the economic instrument with the specific waste materials does not follow from policy intentions of the Dutch government.

2.5 Analytical tools

Rather than using one economic tool to analyze the effectiveness of the different economic instruments shown in Figure 2.2, a range of economic analytical tools is applied. The reason for selecting more than one method is the different characteristics of each instrument considered as well as differences in data availability. Therefore, a tailor-made tool kit was developed to analyze the economic instruments (see Table 2.1). Three of the analytical tools are characterized as quantitative methods: two general equilibrium models WAP I and WAP II, and a partial analysis taking a more sector oriented approach. For instruments and waste flows for which sufficient quantitative data are lacking, a qualitative approach is followed.

Table 2.1 Analytical tools applied in this study

Economic Instrument	WAP I model	WAP II model	Partial analysis	Qualitative analysis
Waste tax changes	Landfill/incineration			Gypsum
Differential waste collection charging		Unit based pricing		
Taxes on resources and products	Paper/card-board/PVC		Sand/gravel	Sand/gravel
Deposit-refund schemes			WEEE/batteries /carpets/mattresses	Carpets/mattresses
Subsidies and public procurement				Food
VAT reduction				Second-hand products

WAP I model

The WAP I model is a general equilibrium model that focuses on the entire economy. Two types of actors are distinguished: households and firms. Households consume goods and supply capital and labour; firms produce goods with the use of capital, labour and intermediate goods. Consumers are differentiated into two types: private consumers and the government. Five different production sectors are distinguished, together producing thirteen unique goods. These sectors are: (1) an extraction sector producing virgin material; (2) a production sector producing eight types of services and goods: 'Wholesale market and auctions', 'Retail market', 'Repairment industry', 'Hotel and Catering industry', 'transport services', 'Financial services', 'other services' and a sector producing all other goods in the economy called: 'rest economy'; (3) a recycling sector producing recycling services; (4) a collection sector producing waste collection services and (5) a waste treatment sector producing incineration services and landfilling services. The hypothetical economy is shown in Figure 2.3.

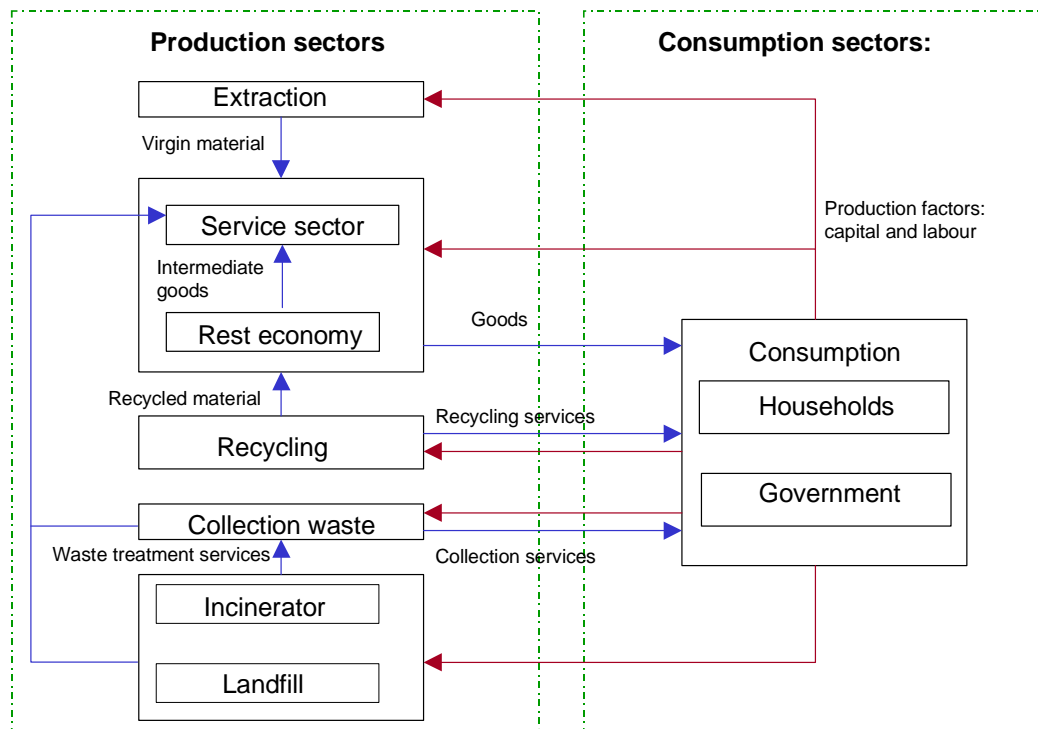


Figure 2.3 Representation of the economy

All firms use capital and labour, which are bought from the consumers, to produce material goods or services. The production sector of consumption goods also uses virgin material or recycled material. Consumption and production lead to waste generation. Producers can recycle waste by investing more capital and labour in the production process. Waste that is generated has to be transported to an incineration plant or landfilling site.

Consumption of goods and services by private households leads to the generation of municipal solid waste. Waste must be either recycled or collected by the municipality. We differentiate two types of municipalities in this model. One municipality charges a flat fee for waste collection; the other municipality charges a unit-based price for waste collection. By including the two types of municipalities we can simulate whether households will react differently to waste policy changes depending on whether they get charged a flat fee or a unit-based price for waste collection.

Consumers can prevent waste by recycling more or, to a lesser extent, by substituting waste extensive goods for waste intensive goods. In reality, consumers have the possibility of two kinds of substitution, namely substitution within a sector and substitution between sectors. Substitution within a sector makes it possible to choose between two products that are basically the same except for waste intensity. Substituting between sectors would mean changing consumption patterns. For example, Linderhof *et al* (2001) show that households in Oostzaan not only bought more products containing less packaging, an example of substitution within a sector, but also began to use diaper services instead of disposable diapers, an example of substitution between sectors after the introduction of unit-based pricing. Waste prevention through substitution within a sector would add a certain degree of complexity to the model, as different products within the same sector and their associated 'waste intensity' would have to be explicitly

modelled. We have chosen to include only the more straightforward channel of waste prevention through substitution between sectors. As a consequence, the possibility of waste prevention may, therefore, be underestimated.

The available incineration capacity in the model will be limited. The total available incineration capacity will be insufficient to fully treat all municipal and production waste. Firms and municipalities, however, will have the option of exporting combustible waste and waste to be landfilled. Thus the limitations of available capacity can be avoided and firms and municipalities will have the option to incinerate all their waste.

WAP II model

The WAP II model is a general equilibrium model like the WAP I model. However, the focus of the WAPII model is more on the consumption sector than the production sector. In the WAPII model, 16 types of households are distinguished. The household types differ in income, household size and type of residence (house/flat). Households will produce 4 types of waste: organic waste, glass, paper, and rest waste. The rest of the economy is simplified to two producers: a producer of a waste intensive good and a producer of a waste extensive good. The WAP II model can be used to determine how specific types of households will react to waste policy changes. Combining this with knowledge about percentages of household types by urbanization degree of the municipalities, we can predict how the production of unit-based pricing will affect waste flows in municipalities with various urbanization degrees.

Partial analysis

For the estimation of impacts of introducing a deposit refund system (DRS), we use a partial general equilibrium model. The theoretical model was developed by Fullerton and Wu (1998). With the Fullerton-Wu model, a number of different environmental taxes and subsidies, such as a deposit-refund system, can be evaluated. In a DRS, products are taxed when consumers purchase the product, and the products are subsidized when they are separately discharged.

The Fullerton-Wu model distinguishes a representative household that consumes commodities and either recycles products or discharges waste. The production of garbage depends on the total amount of consumption goods, and the packaging rate and the degree of recyclability of the commodities. Utility of households depends on the amount of consumer commodities, and the total amount of garbage collected. Households maximize utility subjected to a budget constraint.

Firms produce commodities and have the option to use recycled products or the packaging rate of the commodities. Firms maximize profits. By imposing a tax on packaging of consumer products or consumer products (small electric appliances, batteries, furniture) and a subsidy on recycling, we can evaluate a DRS system.

More details of the Fullerton-Wu model are provided in Appendix III.

Qualitative analysis

In a number of cases, data availability did not allow the use of a (general or partial) equilibrium model to analyse the impact of the economic instrument option. In such

cases, a more qualitative approach was applied, although some quantitative calculations and estimates were made as well. These analyses are mainly based on existing documentation and expert knowledge concerning experiences with, and potential of the instruments under consideration.

3. Waste taxes

3.1 Introduction

Waste taxes are levied on the final disposal of waste; usually on landfilling and (sometimes) incineration. The tax rate can be differentiated according to the type of waste (e.g. homogeneity, density, biodegradability, presence of particular substances) and to particular features of the landfill or the incineration plant. Such differentiations allow some fine-tuning to take into account the environmental impact of the chosen disposal route.

3.2 Theory and literature

From a theoretical point of view, and under certain assumptions, taxes on final waste disposal are the best policy instruments to internalise the environmental externalities caused by waste. Taxes are more efficient than bans or other quantitative restrictions (such as capacity limits), because they leave the choice between final disposal and other options to the market, thus ensuring that for each batch of waste the option with the lowest social cost will be chosen.

To arrive at this social optimum, the tax rates should be based upon (the social value of) the actual damage caused by the disposal option, implying for instance that they should be differentiated according to the emission characteristics of the landfill or incineration plant as well as to the vulnerability of the specific environment where it is sited. In a well functioning market, the price signal from the waste tax will be transferred along the value chain, providing appropriate incentives to all actors to look for alternatives for final disposal, such as prevention, material substitution and recycling.

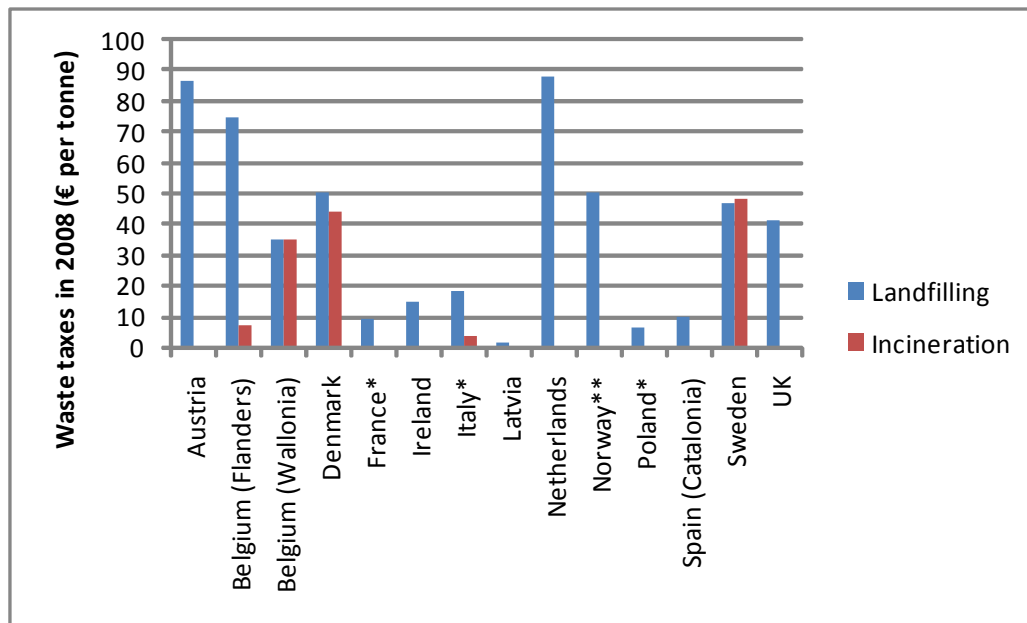
In practice, such theoretical waste taxes do not exist. Tax rates (with the notable exception of the UK Landfill Tax) are not based on environmental externalities and do not always differentiate between waste streams and processing methods according to their environmental impact (see also Appendix I). Moreover, waste markets do not always function well (e.g. in the UK it is impossible for municipalities to transfer the price signal of the landfill tax to households; see e.g. Martin and Scott, 2003), so that the price signal created by a waste tax does not necessarily reach the household or firm that takes the waste disposal decision. Nevertheless, even these ‘imperfect’ waste taxes can play a role in waste policy by making the landfilling and/or incineration option financially less attractive than other options that have a higher rank in the waste hierarchy.

3.3 The Dutch situation

3.3.1 Description

Since 1995 the Netherlands has a tax on the final disposal of waste. This tax is based upon article 23 of the Environmental Taxes Act (‘Wet belastingen op milieugrondslag’). Until now, the area of application has been limited to landfilling, as the rate for

incineration is nil. In 2008, the rate for landfilling amounts to €88.21 per tonne, which is relatively high compared to other EU countries that apply a landfill tax (see Figure 3.1). A limited number of waste categories, including waste with a density of more than 1,100 kg per m³, are subject to a reduced rate of €14.56 per tonne. Dredging sludge is exempted.



* Average of highest and lowest rate.

** Norway also has a tax on waste incineration, but the rates depend on the air pollution characteristics of the incineration plant and could therefore not be shown in the Figure.

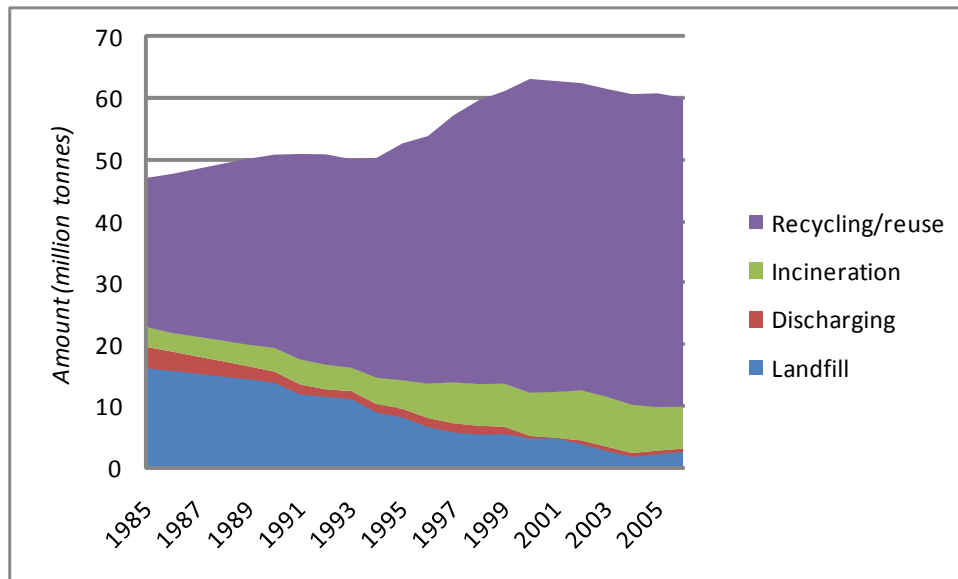
Sources: see Appendix I; for the UK: HM Revenue and Customs.

Figure 3.1 Waste tax rates in selected European countries in 2008 (EUR/tonne)

3.3.2 Impact and effectiveness

An important motive for the introduction of the waste tax in the Netherlands (and for applying a zero rate to incineration) was to bridge the gap between the cost of landfilling and those of other waste treatment options. The landfill tax should make the latter more attractive. In this respect, the landfill tax has been effective: since 2000 the average costs of landfilling (including the tax) have been higher than the costs of incineration (see Bartelings *et al.*, 2005).

The amount of waste sent to landfill has strongly decreased since the introduction of the waste tax (see Figure 3.2). However, this downward trend cannot be attributed to the waste tax alone. First, as Figure 3.2 shows, the decrease was a continuation of a trend that was already present before 1995. Apparently, therefore, already before the introduction of the tax forces were at work that discouraged landfilling. Second, in addition to the landfill tax other policy instruments have had an influence, including the ban (with exemption possibility) on landfilling recyclable and combustible waste, which came into operation in 1995 as well.



Source: SenterNovem/CBS, 2008: www.milieucompendium.nl

Figure 3.2 Waste production and method of treatment in the Netherlands, 1985-2006 (in Mtonnes)

A study on the effectiveness of the waste tax (Bartelings *et al.*, 2005) shows that a direct impact on the amount of landfilled cannot be demonstrated. However, there is a measurable indirect impact (through waste processing fees) on the increase of the amount of waste recycled by companies in the services sector. For households, a similar impact could not be observed, which can be explained by the fact that most municipalities do not (yet) apply differentiated waste collection charges. In those municipalities, higher costs of waste disposal do lead to higher waste collection charges, but this does not provide an incentive to households to reduce their waste supply.

3.3.3 Expectations and prospects

In its Green Paper on market-based instruments for environment and related policy purposes the European Commission (2007) raised the possibility of a certain harmonization of waste taxes in the EU. In this context, it asked: “*If there is insufficient progress to divert waste away from landfill, should the Commission consider proposing a harmonised landfill tax with EU-wide minimum rates?*” The results of the consultation round on the Green Paper (which ended on 31 July 2007) have not yet been published. In its response to the Green Paper, the Dutch government⁴ stated that the Netherlands supports the mandatory introduction of a landfill tax for all Member States, with EU wide ‘environmentally effective’ minimum rates.⁵ In principle, the Netherlands is also in favour of common design criteria for national landfill taxes, provided that differences in circumstances between member States are taken into account.

⁴ Second Chamber of Parliament, 2006–2007, 22 112, nr. 541.

⁵ It does not say what is meant by ‘environmentally effective’.

3.4 Foreign experiences⁶

Several countries in the EU and elsewhere apply waste taxes. Figure 3.1 shows the (standard) tax rates for (combustible) municipal waste in a number of European countries. In several countries the waste tax is restricted to landfilling. Presently, Belgium, Denmark, Italy and Sweden are the only EU countries that apply it to incineration as well. In Belgium (Walloon region) and Italy only incineration without energy recovery is taxed. Sweden has a reduced rate for incineration with both heat and power generation. Denmark and the Flanders region in Belgium apply a single rate to all types of waste incineration. A proposal to introduce a tax on waste incineration has been put forward in France.

Differentiations in landfill tax rates between types of waste are quite common, e.g. between combustible and non-combustible waste, between single and mixed waste streams, and between hazardous and non-hazardous waste.

Some countries, apply lower rates to landfill sites that meet certain environmental criteria. In Austria, Estonia and Norway these criteria are related to the presence of environmental protection measures. In France, landfills with an environmental management system (EMAS or ISO 14000 certified) qualify for a reduced rate.

The waste tax in Wallonia is only levied on communities where the average amount of municipal waste exceeds a certain threshold (240 kg per inhabitant per year).

Revenues from waste taxes generally accrue to the general budget, but several countries earmark them for specific waste related or other environmental purposes. This is common practice in the new Central and Eastern European Member States, but waste tax revenues are also earmarked in Austria, Ireland, Italy and Spain (Catalonia).

Waste taxes are always part of a package of waste policy instruments and it is therefore hard to disentangle their impact (on waste reduction, recycling and diversion to alternative ways of final disposal) from the impact of other instruments. Evaluations of landfill taxes in the UK (Ecotec, 2001) and Denmark (Andersen, 1998) suggest that these taxes led to significant reductions in the landfilling of construction and demolition waste, whereas other impacts were less pronounced or even absent. Austria reports positive effects of the rate differentiation between landfill sites that meet certain environmental standards and those that do not (Umweltbundesamt, 2000).

3.5 Case study

The following options will be analysed with respect to taxes on the final disposal of waste (landfilling and incineration):

1. Differentiation of the landfill tax rates (changes in the absolute and relative values of the standard and the reduced rate) including adaptations of the waste categories qualifying for the reduced landfill tax rate;
2. Introduction of a positive tax rate for incineration, with or without differentiation by energy efficiency of the incineration plant.

⁶ For details by country, see Appendix I.

Under option 1, we will look specifically at the possibility to transfer a specific waste category from the reduced rate of the waste tax to the standard rate. Currently, waste with a density of more than 1100 kg per m³ (mainly construction and demolition waste, CDW) is taxed at the reduced rate. Gypsum, which constitutes around 2% of CDW on average, has been identified as a priority waste stream. The analysis will therefore focus on applying the standard rate to CDW with a gypsum content of more than 1%. This might create a stimulus to separate gypsum from the remaining CDW.

Option 2 has been selected in order to analyse whether a tax on waste incineration might stimulate the choice of alternatives that rank higher in the 'waste hierarchy', such as prevention and recycling. A possible differentiation in tax rates according to energy efficiency of the incineration plant could possibly contribute to a better use of the energy incorporated in the waste, both in terms of electricity and heat.

3.5.1 Gypsum in construction and demolition waste⁷

During the last decades, gypsum has been increasingly used as a building material, although the growth recently seems to have come to an end. This implies that the amount of gypsum in CDW will continue to grow for the next 40 years. The presence of gypsum in CDW hampers its useful application in for instance road construction, due to its sulphate content. Separating the gypsum from CDW will therefore not only facilitate the recycling of the gypsum itself, but also the recycling of the remaining CDW.

The current use of gypsum in the Netherlands is around 1 million tonnes per year. About half of this consists of gypsum panels and blocks; other applications are gypsum plaster, anhydrite floors, and cement. A substantial part of the gypsum used is a by-product of electricity production: it is produced during flue gas desulphurisation (FGD) in coal fired power plants.

CDW contains, on average, some 2% of gypsum. Half of this is 'visible' gypsum from panels and blocks; the other half is 'diffuse' or 'invisible' gypsum from plaster and mortar. The total amount of gypsum in CDW is estimated at about 100 Ktonnes per year, although this figure has rather wide uncertainty margins. According to the most recent TNO figures, in 2007-2008 32 Ktonnes of gypsum were separated from the other CDW, of which 17 to 22 Ktonnes were recycled in two plants in the Netherlands. The remainder (between 7 and 15 Ktonnes) was exported, mainly to Germany, where it is used (among others) as covering material for landfills and filling for abandoned mines. What happens with the unseparated gypsum (mixed with other CDW) is largely unknown, although it is likely that a substantial part is also exported.

The export of gypsum is considered to be environmentally inferior to recycling or even landfilling, mainly due to the additional emissions from transport (CE, 2008).

The option under consideration here is to apply the 'high' rate in the waste tax (in 2008: €88.21 per tonne) to the landfilling of waste containing more than 1% by weight of gypsum. Presently, CDW is taxed at the reduced rate of €14.56 per tonne. The primary impact of this measure would be an increase in the costs of landfilling CDW with a high gypsum content by €74 per tonne. This may create a price incentive stimulating the

⁷ For this subsection data were kindly provided by TNO (ms E.M.G. Roelofs).

separation of ('visible') gypsum from the remaining CDW. TNO estimates that this separation leads to a 20-25% increase in the labour costs of demolition. In 2004, the cost of landfilling unsorted CDW was about €140 per tonne; whereas the price charged for collecting 'pure' gypsum waste by a specialised firm was around €45 per tonne.⁸ In this situation, separation was already profitable in some cases. The higher waste tax rate would enlarge this price difference from €95 to almost €170 per tonne. This is a significant increase and it is likely that it would make separation of gypsum from CDW financially attractive in many more cases.

Obviously, this will only happen if there are no other, cheaper ways to dispose of the CDW with high gypsum content. Its use in e.g. road construction is probably not allowed (due to the presence of sulphate), but export of unsorted CDW for landfilling abroad may become (or remain) an attractive option after the introduction of a higher waste tax in the Netherlands.

The environmental impact of better gypsum separation is uncertain, as it depends to a large extent on the destination of the separated gypsum. If this is exported (and thus transported over a long distance) and used for 'low value' purposes (such as mine filling), the net environmental gains may be small or even negative. If the gypsum is processed and recycled near to the place where it originated, it is much more likely that there will be positive environmental benefits. Theoretically, an increase in the supply of recycled gypsum could oust the existing use of FGD gypsum, which itself is a useful application of a waste stream. However, TNO expects that the in the foreseeable future an oversupply of FGD gypsum on the European market is unlikely.

To summarize, introducing a high waste tax rate for CDW with a gypsum content above 1% might provide a strong incentive for more separation of gypsum from other CDW, but its relevance remains unclear due to insufficient data on landfilling and export. The environmental merit of this measure will depend on the destination of the separated gypsum. In order to stimulate gypsum recycling, it may be necessary to introduce additional measures that make other options less attractive or impossible. Furthermore, for some recycling purposes the purity of the gypsum is very important, and therefore a quality assurance system may be needed.

3.5.2 Incineration and landfill taxation

As noted before, the landfill tax has a high 'standard' rate, and a reduced rate for some categories, including waste with a density of more than 1100 kg per m³ (which is mainly non-combustible waste). Table 3.1 shows that the high rate started at a fairly low level and has been increased substantially over the last decade. Especially in 2000, the high rate was increased substantially, so that the price of incineration became lower than the price of landfilling. Thus far the tax rate for incineration has been zero. Introducing a positive incineration tax rate without raising the landfilling tax will again raise the price of incineration above landfilling thus providing municipalities and the HDO⁹ sector with

⁸ Excluding collection and transport costs. This price was charged for gypsum to be exported to Germany. The costs of selected recycling processes were estimated to be in the same order of magnitude.

⁹ HDO is the abbreviation of the Dutch term for Trade, Services and Government.

an incentive to landfill more waste. In the case study presented in this subsection only the high rate will be used as household waste and waste of the service sector hardly contains any non-combustible waste.

Table 3.1 Landfill tax (in € per tonne)

Type of waste	1996-1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008
Combustible waste	29.1	30.3	64.3	65.4	78.8	81.7	83.6	84.8	85.5	86.9	88.2
Non-combustible waste	13.3	13.3	12.4	12.6	13	13.5	13.8	14	14.1	14.3	14.6

Source: Statistics Netherlands (CBS); 'Wet belastingen op milieugrondslag' (Environmental taxes act), art. 18.

To simulate the effects of taxation on waste generation and treatment, three scenarios are considered (see Table 3.2). In the first scenario the landfill tax is increased to a maximum of €100 per tonne of waste landfilled. In the second scenario a tax of €40 per tonne is introduced on the incineration of waste and the landfilling tax is kept at €80 per tonne. This will effectively raise the price of incineration over the price of landfilling. In the third scenario both the higher landfilling tax of 100 euro per tonne and the incineration tax of €40 per tonne are introduced.

Table 3.2 Taxation scenarios for landfilling and incineration

Scenario	Landfill tax (€/tonne)	Incineration tax (€/tonne)	Result
1	100	0	Landfill costs exceed incineration costs
2	80	40	Incineration costs exceed landfill costs
3	100	40	Both waste treatment options become more expensive

The main outcome of these scenarios is presented in Figure 3.3 and Figure 3.4 for household waste and HDO waste, respectively. Figure 3.3 shows the effects that can be expected for the manner in which municipal solid waste is processed in the Netherlands. None of the scenarios show a significant change in the total amount of waste generated. Introducing an incineration tax leads to slightly more landfilling of solid waste due to the lower cost of landfilling compared to incineration. However, since municipalities have little flexibility in switching to landfilling, due to the landfilling ban and long term contracts between municipalities and incinerators¹⁰, only a minor effect is detected.

¹⁰ The landfilling ban and long term contracts are simulated in the WAPI model by inelastic substitution elasticity between landfilling and incineration.

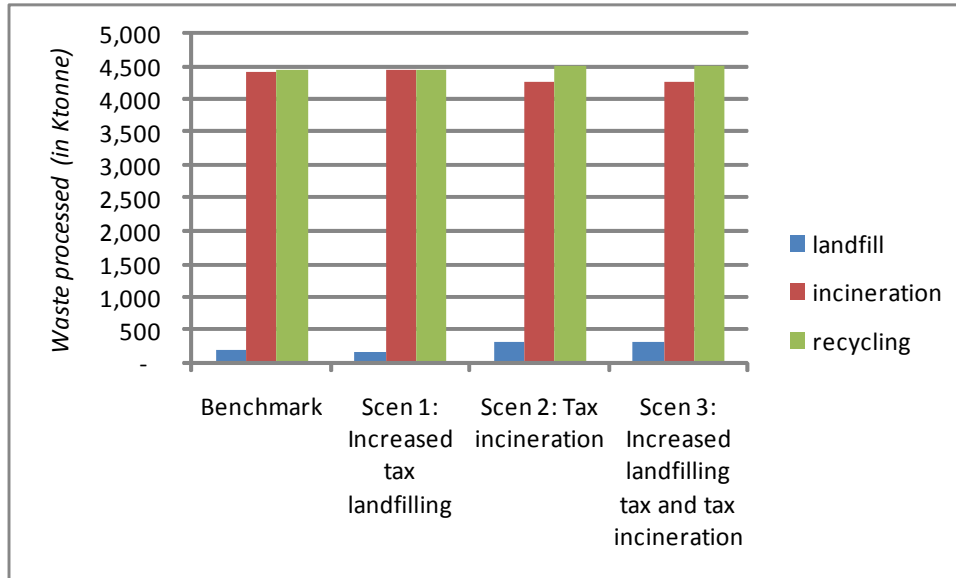


Figure 3.3 Amount of solid waste generated by households landfilled, incinerated or recycled

The extent to which taxes lead to changes is also determined by the ability of municipalities to pass on the costs to the consumers. For example, municipalities with unit-based pricing can charge a higher price for waste collection because of the extra cost of incineration and/or landfilling. As a result, households in those municipalities will increase waste separation that in turn results in a slight increase in recycling of waste in these municipalities. However, the overall effect of this mechanism is quite small because only a limited number of municipalities currently charge a unit-based price for waste collection and the increase of the price for waste collection due to tax increases is relatively small. In the municipalities charging a flat rate for waste collection, the link between the costs of waste treatment and the amount of waste generated does not exist and in these municipalities the amount of waste separated will not change as a result of higher waste treatment costs.

The results for the generation of (service) waste in the HDO sector are shown in Figure 3.4. As the HDO sector has a more direct link between the costs of waste treatment and the generation of waste, potentially a higher tax on waste treatment could increase recycling. Figure 3.4 shows that recycling increases, however, since the taxation in all three scenarios is not high enough to offset the relatively higher costs of recycling, the increase in recycling is only marginal. Introducing an incineration tax leads to a situation where more service waste is landfilled due to the lower costs of landfilling relative to incineration in Scenario 2 and 3. To demonstrate how the tax rate can influence waste treatment more significantly, a sensitivity analysis was conducted in which the tax rate was increased subsequently from zero to €200 per tonne in the case of landfilling and from zero to €100 in the case of incineration.

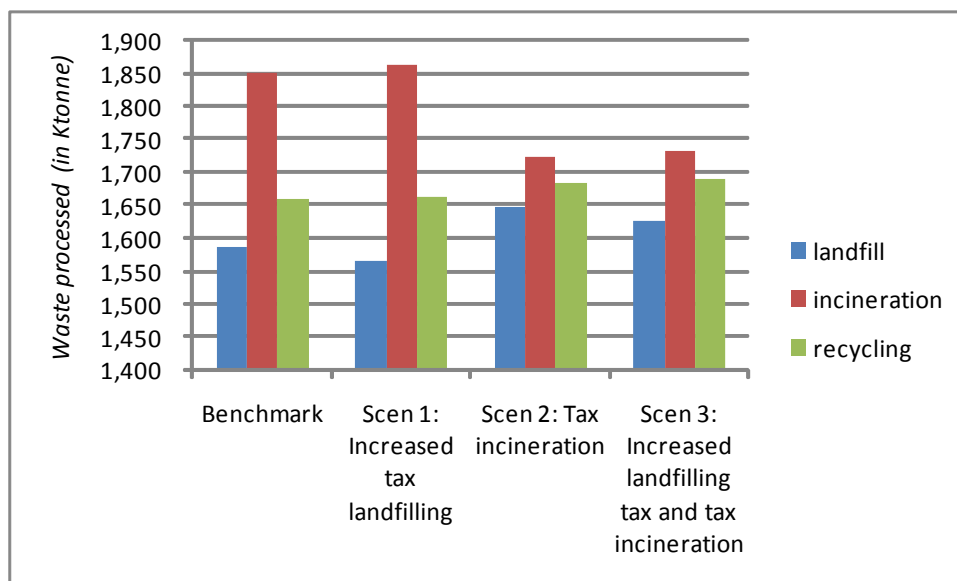


Figure 3.4 Amount of solid waste generated by the service sector landfilled, incinerated or recycled

Figure 3.5 shows how the total amount of waste landfilled, incinerated or recycled is affected by the landfill tax rate. The marginal benefits of the landfill tax are especially high in the beginning. Going from a low landfill tax rate to a slightly higher landfill tax results in a substantial decrease in waste landfilled. The amount of waste incinerated does not increase as much as the amount of waste landfilled decreases, indicating that households and industries do increase recycling. Due to the limited capacity of incineration plants, incineration cannot increase beyond a certain level. Given a landfill tax of about €100 per tonne, the complete incineration capacity is utilised and therefore incineration is unable to increase further.¹¹

The difference between household behaviour and industries in responding to changes in the landfill tax is quite noticeable (not shown in the figure). Municipalities, who collect household waste, switch from landfilling to incineration. If the landfill tax is very high, almost all-household waste will be incinerated. The service sector also increases incineration and decreases the amount of waste landfilled. However, if the tax exceeds €100 per tonne all household waste will be incinerated. Due to the limited capacity of incinerators, the remaining incineration capacity is too small to incinerate all the service waste as well. As a result, the service sector is forced to start landfilling the waste again. Recycling in the service sector also increases slightly, however the costs of increasing recycling is quite high thus making the effect only limited.

¹¹ To represent the public opposition against having an incinerator “in your backyard” and the high capital costs and long construction period involved in building new incinerators, the model does not allow for the option of increasing incineration capacity.

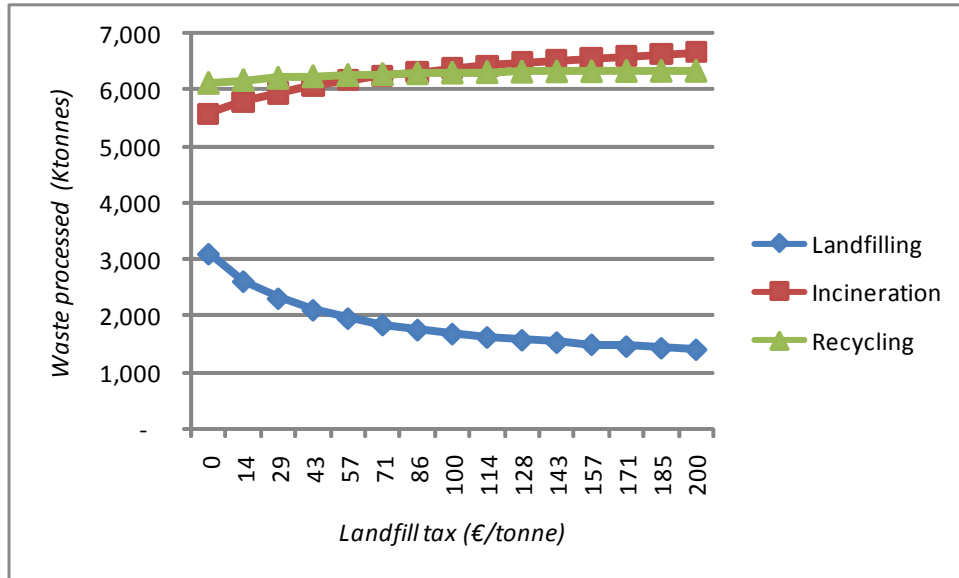


Figure 3.5 Waste treatment and the landfill tax (with incineration tax rate zero)

The next sensitivity analysis involves the increase of the incineration tax zero to €100 per tonne. Figure 3.6 shows how the total amount of waste landfilled or incinerated is affected by the incineration tax rate. Increasing the incineration tax influences mainly the choice between incineration and landfilling. Landfilling becomes more financially attractive again and as a result mainly the service sector, which is more responsive to price changes, will aim at landfilling more waste. Because landfilling becomes the most attractive and cheapest option, recycling will hardly increase as a result of an increase of the incineration tax.

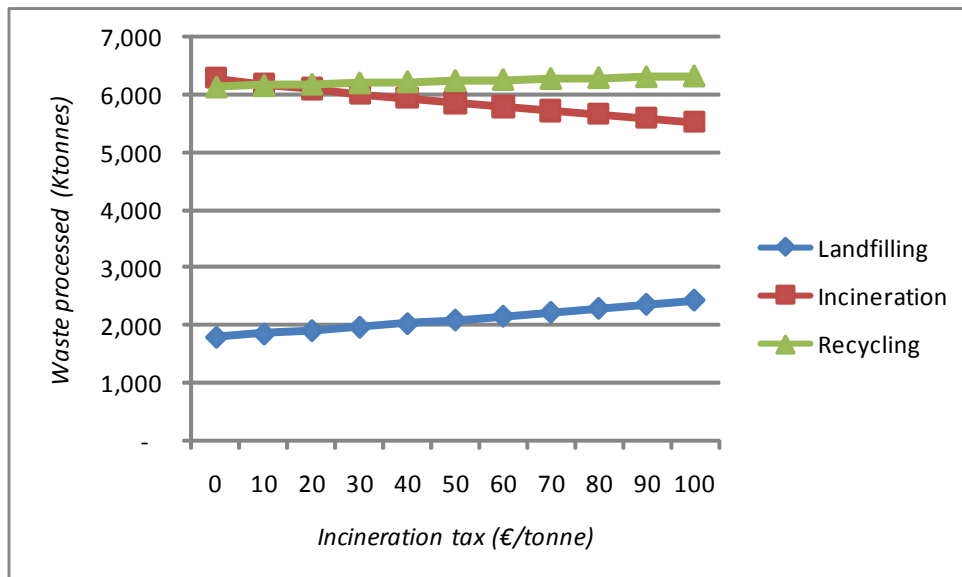


Figure 3.6 Waste treatment and the incineration tax (with landfill tax rate zero)

Figure 3.7 shows how the total amount of waste landfilled or incinerated is affected by an increase in both the incineration tax rate and the landfilling tax rate. It is assumed in this sensitivity analysis that both the tax rate for incineration and landfilling start at the

level of 2005 (zero and €80 per tonne, respectively). Increasing both tax rates basically only affects the relative cost of recycling, which becomes lower. The service sector recycles slightly more. However, since most households do not experience a direct link between cost of waste treatment and generation of waste, they are not affected by the measure and do not recycle more as a result of higher taxes.

The substitution elasticities for the service sector are based on information about the actual amount of waste recycled and the costs of waste treatment in the period 1995-2003 (see Bartelings *et al*, 2005). These substitution elasticities are fairly inelastic thus constraining the switch between recycling and waste treatment. This may explain the relatively small increase in the amount of waste recycled by the service sector despite the fact that this sector is perceived to have a direct price incentive to recycle more. The inelastic substitution elasticities are for a large part explained by the fact that many of the firms operating in the service sector are not really very different from the households in terms of their manner of waste disposal. Municipalities collect waste for many of these firms and they may or may not charge a variable price for this. Therefore, also for the service sector there may not be a direct price incentive to reduce waste when waste treatment costs increase.

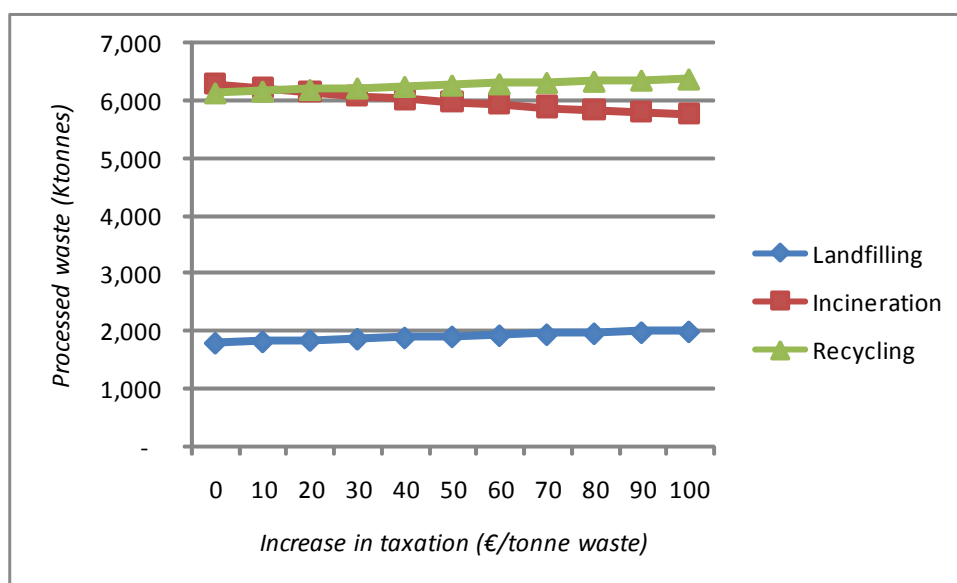


Figure 3.7 Waste treatment and both the incineration and landfilling tax (with landfill tax rate €80 per tonne higher than the incineration tax rate)

The model does not take into account differentiation by energy efficiency of the incineration plant. Table 3.3 shows the energy efficiency of the incinerators in the Netherlands, revealing a noticeable difference between the incinerators. Given the current limitations on incineration capacity, the effect of differentiation in taxation by energy efficiency will be limited. Such a differentiation will only work if the energy efficient incinerators have enough capacity to incinerate more waste. As long as they are already operating at full capacity, both municipalities and the production sector will not have the option of switching to more energy efficient incinerators. Therefore, the differentiation in taxation on the basis of energy efficiency will not be effective in the Netherlands. Moreover, it should be noted that municipalities in the Netherlands have

long standing contracts with incinerators. Any effect of taxation may therefore take a while to generate an effect because renegotiations of these contracts and changes in behaviour are only possible after the contracts have ended.

The model is also limited to a closed economy, which means that import and export of waste is not permitted. In theory, introducing a tax on incineration may well lead to a situation in which more waste is exported. The extent to which export increases depends on the availability of processing capacity in the importing country, the international transport costs and the level of the tax set in the exporting country. Export of waste will diminish the impact of an incineration tax on the level of recycling. Since municipalities and the service sector have an alternative option to cheaply dispose of their waste, the incentive to increase recycling will be even smaller. In reality, however, the impact of international trade on the efficiency of a tax on incineration will be limited: the incineration capacity of neighboring countries is also constrained and Dutch municipalities often maintain long-term contracts which limit their flexibility in exporting combustible waste.

Table 3.3 Waste incineration plants in the Netherlands with energy efficiency rating

Incinerator	Waste treatment 2006 (kton)	Net energy efficiency	Sustainable energy (GWh)	Sustainable heat (TJ)	Avoided use of fossil fuels (TJ)
AVR Rozenburg	1,170	0.25	204	87	2,594
AEB Amsterdam	943	0.23	254	101	2,197
AZN Moerdijk	670	0.56		2,664	2,936
HVC Alkmaar	640	0.23	189	2	1,410
Essent Wijster	556	0.20	134		1,008
AVR Rotterdam	385	0.13	59		483
AVR Duiven	345	0.23	56	249	714
Twence Hengelo	289	0.17	68		509
ARN Weurt	270	0.29	62	318	813
Gevudo Dordrecht	218	0.07	19		102
Sita Dordrecht	58	0.11		39	28
Total	5,544	0.26	1,045	4,245	12,794

Source: SenterNovem (2008a).

As shown in Figure 3.8, introducing (higher) taxes on landfilling and incineration has negative welfare effects, be it on a relatively small scale. Note that the welfare effects as presented in Figure 3.8 exclude possible environmental welfare effects. Introducing both a higher landfilling tax and an incineration tax has the largest effect of welfare as it increases the household cost of waste treatment, thus inflating the price of goods and reducing consumption slightly. Increasing the landfill tax hardly affects welfare: since the amount of waste landfilled in the benchmark scenario is already small, higher landfilling costs hardly affect the way waste is treated.

Since the amount of waste recycled, landfilled or incinerated hardly changes in the three scenarios, the environmental impact of the introduction of incineration taxes is also limited (see Table 3.4).

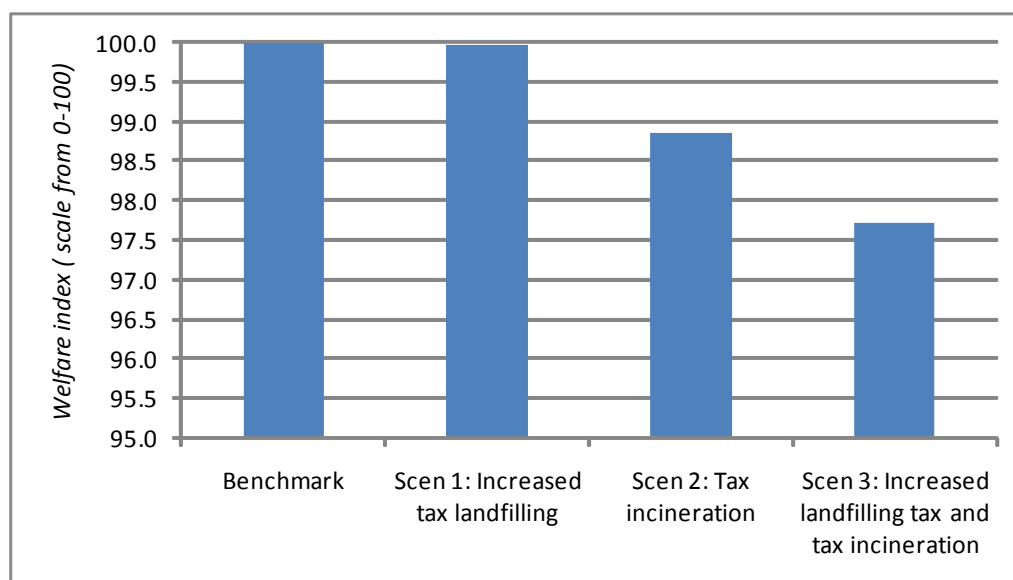


Figure 3.8 Impact on welfare (indexed on a scale from 0-100)

Table 3.4 Impact of tax on environment (benchmark = 100)

Environmental indicator	Scen 1: Increased tax landfilling	Scen 2: Tax incineration	Scen 3: Increased landfilling tax and tax incineration
Total amount of waste	99,90	99,09	98,48
Abiotic exhaustion (ADP)	99,84	97,91	98,39
Climate change (GWP100)	99,91	99,24	98,49
Ozone layer (ODP)	99,84	97,99	98,40
Human toxicity (HTP)	99,78	96,81	98,30
Smog formation (POCP)	99,86	98,21	98,41
Energy use (MJprim)	99,91	99,21	98,49

3.6 Conclusions

Waste taxes are economic instruments interfering at the very end of the chain: the point of final disposal. This implies that their effectiveness will depend on whether or not their incentives will be transferred to the preceding links in the chain. In other words, waste taxes will only have the desired impact if the resulting higher costs of landfilling and incineration effectively make waste prevention and recycling financially more attractive for the various actors. For this to happen, the simultaneous presence of other instruments, such as differential waste collection charge rates (see next Chapter 4), is required.

In the present situation in the Netherlands, the landfill tax of €88 per tonne makes incineration (which is not taxed at all) a financially more attractive option than landfilling. Introducing taxes on incineration and higher landfilling taxes without other instruments is hardly effective. The two main reasons for this inefficiency are the presence of ample market imperfections in the current waste market and the lack of price incentives for the producers of waste to recycle more. A higher landfill tax will hardly have an effect on the amount of municipal waste incinerated as almost all the household waste is already incinerated and the capacity of the incineration is not large enough to also incinerate additional waste from the service sector. Introducing a positive

incineration tax rate while holding the rate for landfilling constant will effectively raise the price of incineration over the price of landfilling again, thus making it more attractive to landfill waste. Combining an incineration tax and a high landfilling tax can potentially stimulate recycling. However, since households are hardly affected by these taxes, they will not take much more effort to separate their waste for recycling purposes. The service sector is somewhat more sensitive for changes in tax. However, since the cost of increasing recycling is quite high, based on data from 1995-2003, the landfill and/or incineration tax will need to be considerable to see a notable increase in recycling in this sector.

Differentiating the tax on incineration according to the energy efficiency of the incinerator may actually stimulate municipalities to use more energy efficient plants. However, in the short term a severely limiting factor to this measure is the maximum capacity of the incinerator. If the energy efficient incinerator is already used to its full capacity, as is almost the case in many regions in the Netherlands, municipalities will not have the option of switching to that incinerator and thus the measure will be ineffective. There are no clear signs in the Dutch waste sector indicating the incineration capacity in the Netherlands to expand. Besides the limited capacity, the flexibility in household waste management practises is constrained by the existence of long-term contracts between waste processors and municipalities.

It is crucial that simultaneous to the consideration of taxes on waste treatment, measures are taken which transfer the incentives of the tax on to the producers of waste (households and the service sector). This can be done for example by introducing unit-based pricing as is discussed in the next chapter. Besides that, it is important to keep in mind that there should be enough capacity to switch to the most desirable waste treatment option.

4. Waste collection charges

4.1 Introduction

The collection of municipal waste from private households is usually carried out by, or on behalf of, municipal/local authorities. In most countries, the costs of this activity are covered by waste collection charges (a notable exception is the UK, where such charges are not allowed and financing is through the general budget of the local authority). These charges can be used as economic incentives by relating them as closely as possible to the amount of waste supplied and the (potential) environmental harmfulness of the waste. In other words: the polluter pays, and the heaviest polluters pay the most.

4.2 Theory and literature

In principle, the use of differential and variable rates (DVR) in waste collection charging can be seen as a necessary complement to waste taxes, in order to ensure that the price incentive of the latter is transmitted to households and thus contributes to waste reduction and recycling. As waste collection is usually a public service, the usually recommended approach in the literature is to set the price that is to be charged equal to the long-run marginal cost of providing the service. This sets an incentive to householders not to over consume the service while on the other hand forcing bureaucrats away from inefficient allocations due to (among others) greater cost transparency (Morgenroth, 2006).

Early evidence on the effectiveness of DVR was presented by Kinnaman and Fullerton (1997). More recently, the OECD (2006a) has investigated the costs and benefits of DVR in waste collection charging. A literature review revealed, among others, that weight-based and pay-per-bag schemes appear to perform best, while schemes for which the charge is based upon the volume of the container perform worst (see for example Dijkgraaf, 2004). Furthermore, schemes are likely to perform best where there is a comprehensive system in place for collection of segregated materials for recycling. The effects of DVR on illegal waste disposal are difficult to estimate.

The same OECD (2006a) study used case studies (in Belgium, Germany and Spain) to arrive at a social cost-benefit estimate for DVR. A key conclusion was that in each of the cases the net social benefit was positive across most of the scenarios that were analysed. Only where external costs of air pollutants as well as the avoided costs of treatment/disposal were assumed to be low, did the balance of costs and benefits result in a net cost associated with applying the DVR scheme.

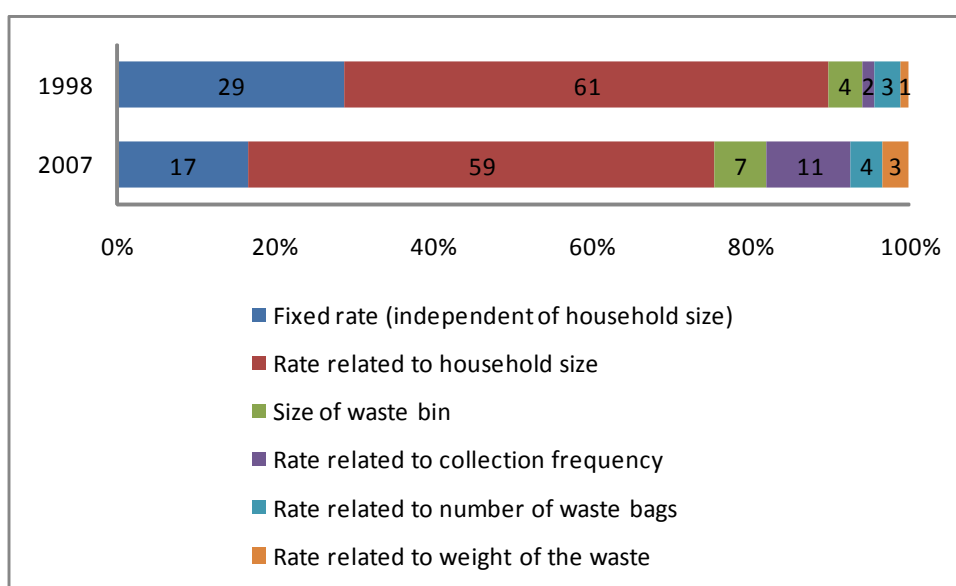
4.3 The Dutch situation

4.3.1 Description

The Environmental Protection Act ('Wet milieubeheer', Wm), article 15.33, authorizes municipalities to levy a charge to cover the expenses they incur in the management of household waste. At present, all municipalities in the Netherlands use this opportunity to

levy a waste collection charge.¹² In 2007, the average charge rate for a household with more than 1 person amounted to €263 (COELO, 2007). In 83% of the municipalities the charge covered the full costs of waste collection.

Municipalities are free to determine the charge base.¹³ Currently, over 30% of them (especially relatively small municipalities outside the Randstad region) apply a rate that depends on the amount of waste supplied (SenterNovem, 2007). This kind of charge rate setting (referred to as DVR in the previous section) is known in the Netherlands as ‘diftar’ (an abbreviation of the Dutch term for ‘differentiated rates’: ‘gedifferentieerde tarieven’). In 2007, 17.9 percent of the Dutch population lived in a ‘diftar’ municipality; in 2000 this was 9.7% (COELO, 2007).¹⁴ Figure 4.1 shows the growth in the share of different types of ‘diftar’ schemes, and Figure 4.2 the growth in ‘diftar’ schemes among Dutch municipalities.



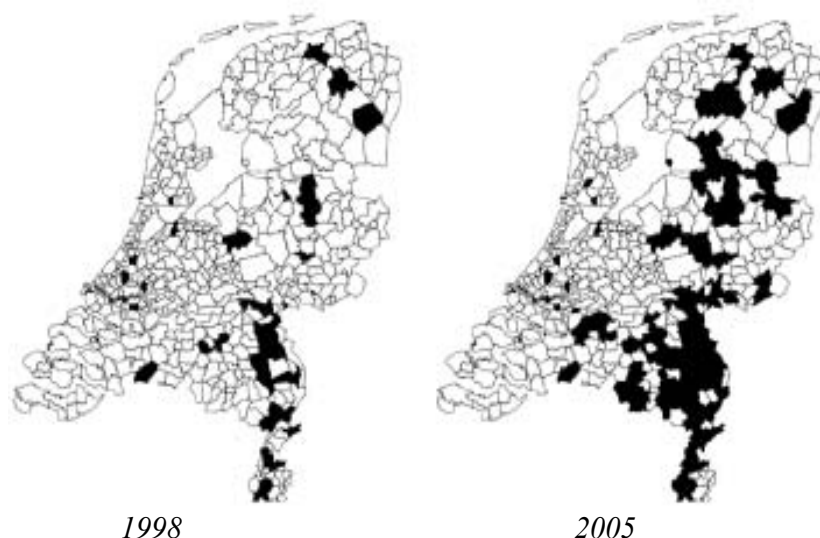
Source: COELO, 2007

Figure 4.1 Waste collection charging scheme types, 1998 and 2007

¹² Two municipalities (Leiden and Eemsmond) have applied a zero rate during some years, but since 2007 they have returned to a positive rate. An unknown (but probably small) number of municipalities does not apply the Wm based waste collection charge, but applies instead a ‘sanitation levy’ (‘reinigingsrecht’, a retribution based upon article 229 of the Municipalities Act (Gemeentewet)). Sanitation levies can only be levied from those who actually use the municipality’s waste collection services, whereas the Wm charge can be levied from all those who reside in an area where the municipality has the obligation to collect waste. In the remainder of this section, the term ‘waste collection charge’ is used for both types of charges.

¹³ However, municipal taxes may not be dependent on income, profit or wealth (article 219 of the Communities Act).

¹⁴ In contrast with SenterNovem, COELO does not count charging based on container volume as a ‘diftar’ system.



Source: Heijnen, 2007.

Figure 4.2 'Diftar' schemes by municipality, 1998 and 2005

4.3.2 Impact and effectiveness

The impact of rate differentiation depends on the type of rate setting chosen (COELO, 2007; Dijkgraaf and Gradus, 2004). Payment by weight or by number of bags roughly leads to a decrease in waste supply of 60% for vegetable and garden waste, and of 50% for other (unsorted) waste. This is mainly caused by a large increase in the amount of waste that is separated by the households (paper glass, etcetera). Obviously, such results will only be achieved if the introduction of DVR is accompanied by the provision of adequate capacity and facilities for separate collection.

Remarkably, a 'bag' system is generally as effective as a 'weight' system, whereas the latter is much more expensive. Payment systems based on frequency of collection lead to reductions of 35% for vegetable and garden waste, and 25% for other (unsorted) waste. Charging rates linked to the volume of the waste bin appears to have hardly any impact on the amount of waste offered.

One of the reasons why not all municipalities apply a 'diftar' scheme is the risk of 'waste tourism': the phenomenon that garbage is dumped (fly-tipping) or supplied through other channels (e.g. family, work). Obviously, the opportunities for 'waste tourism' are reduced when neighbouring municipalities also use 'diftar'; this explains the regional clusters of 'diftar' schemes (COELO, 2007). Municipalities that have introduced 'diftar' do not report any significant increase in littering. Anyway, the additional costs of removing any 'diftar' related litter are much lower than the savings that a 'diftar' system brings with it (CE, 2004). Though the introduction of 'diftar' may initially lead to higher costs, these appear to be earned back due to the decrease in waste supplied (COELO, 2007; Linderhof *et al.*, 2001).¹⁵

¹⁵ In the case of the Oostzaan municipality, the waste *collection* costs after the introduction of 'diftar' increased from NLG 664,000 in 1992 to NLG 805,000 in 1996. The waste *treatment* costs (vegetable and garden waste, and non-recyclable waste) decreased in the same period from NLG 364,000 to NLG 161,000 (Linderhof *et al.*, 2001, Table 5).

4.3.3 Expectations and prospects

Although the number of ‘diftar’ municipalities is still increasing, it is often assumed that the system is not suited for large towns and cities. Nevertheless, there are some urban municipalities (over 100,000 inhabitants) applying some kind of ‘diftar’, such as Nijmegen, Maastricht and Apeldoorn (COELO, 2007). However, as far as the present authors know, there are currently no plans to introduce any kind of ‘diftar’ in the big cities in the Randstad region. It can therefore be expected that for the time being the majority of Dutch households will not pay in proportion to the amount of waste they supply.

Among those municipalities that do apply ‘diftar’, a further increase in system variety is conceivable. For instance, rates could be differentiated by type of waste. Until now, most Dutch municipalities use a uniform rate for all types of waste. Countries like Belgium, Denmark and Switzerland, where a lower (or zero) rate is applied to vegetable and garden waste, have seen a growth in the share of this waste type offered separately (CE, 2004; see also section 0). Applying lower rates to separated waste streams obviously increases the risk that these streams will be contaminated by the ‘rest waste’ that is charged at a higher rate, but quantitative evidence on this phenomenon is lacking.

4.4 Foreign experiences¹⁶

Local authorities in several other countries apply DVR (‘diftar’) systems to some degree. In Belgium, it is much more common than in the Netherlands: almost all municipalities in Flanders and about 70% of the municipalities in Wallonia apply some kind of differentiated waste collection charges.

In South Korea, DVR has been mandatory for all municipalities since 1995. A ‘pay-per-bag’ system is applied nation-wide, with the municipalities deciding on the specifics of the bag and its price. The revenues should cover the costs of accompanying (free) collection services for recyclable and compostable waste. The Korean scheme is reported to be quite successful, leading to substantial growth in recycling activities and in waste prevention initiatives. Illegal disposal was found to be a problem following the introduction in 1995, but has been reduced drastically by means of an enforcement programme.

In the USA, DVR is mandatory for all municipalities in the state of Minnesota. In Europe, no country has yet taken the step towards mandatory DVR although France, Ireland and Italy have expressed their intention to do so. On the other hand, some European countries (such as the UK and Greece) do not even allow their municipalities to apply DVR.

4.5 Case study

As shown in Table 4.1, the implementation of DVR in the Netherlands declines with the urbanization degree of municipalities. In very strongly urbanized municipalities, there is currently no DVR being applied. Forty-four percent of the marginally urbanised municipalities have some kind of DVR system implemented. This DVR coverage share is somewhat lower for non-urbanized areas, implying a discouraging role of higher costs

¹⁶ For details by country, see Appendix I.

of transportation. Overall, 32% of the Dutch municipalities ran some kind of DVR system in 2006, implying that 16% of the population in the Netherlands operates in a DVR setting. Table 4.2 provides further details of the type of charging systems operated in the municipalities in the Netherlands.

Table 4.1 Share of municipalities with DVR in 2006

Urbanization degree	Number of municipalities	Number of inhabitants (x 1 million)	Share of municipalities with DVR (%)
Very strongly urbanized	12	2.90	0
Strongly urbanized	57	4.35	8.8
Moderately urbanized	89	3.48	20.2
Marginally urbanized	160	3.48	44.4
Not urbanized	140	2.09	36.4
Netherlands	458	16.31	31.7

Source: SenterNovem 2008a

Table 4.2 Implementation of DVR systems in the Netherlands, categorised by level of urbanization, 2006

	Very strongly urbanized	Strongly urbanized	Moderately urbanized	Marginally urbanized	Not urbanized
<i>No DVR charge system</i>					
Household size	8	37	54	80	83
Fixed charge	3	15	17	6	5
Other system	0	0	0	3	0
No charge	1	0	0	0	1
<i>DVR system</i>					
Weigh-based	0	0	1	10	9
Weight-based/frequency	0	0	1	3	2
Volume	0	1	6	15	10
Volume/frequency	0	1	8	30	26
Household size/volume/frequency	0	0	0	1	0
Pay per bag	0	2	2	6	1
Pay per bag/household size	0	1	0	5	3
Other types of DVR	0	0	0	1	0
Total number of municipalities	12	57	89	160	140
Share of municipalities	100%	91%	80%	56%	64%
Share of inhabitants	0%	9%	20%	44%	36%

Source: SenterNovem 2008a

Figure 4.3 shows the average waste production for six waste flows of household waste. Rest waste increases with urbanization rate, while organic waste, recycled paper and recycled glass decline with urbanization rate. In very strongly urbanized municipalities the rest waste amounts 320 kilograms per person per year. This is two-third of the total waste. The total amount of waste for inhabitant of not urbanized areas is more than 490 kilograms per year, of which a relatively large share is comprised of organic waste.

Given the experiences abroad, there does not seem to be any *a priori* reason why the penetration of DVR in the Netherlands could not be much higher than the current level. We will therefore analyse the impact of some options for a more extensive use of DVR systems in the Netherlands.¹⁷

For the analysis, we focus attention on four waste streams. Large rest waste and textiles are excluded. Large rest waste is very typical type of waste, because the alternatives for waste disposal are quite different. Alternatives are second-hand shops and the virtual second-hand market via the Internet. Textiles are not included, because the waste flow is rather small.

The analysis of increasing the penetration of DVR in the Netherlands is based on the urbanization degree of the municipalities. Since DVR is more likely to be implemented in the less urbanized municipalities, the scenarios presented take into account the urbanization degrees of municipalities. For the analysis, we use the WAP II model, in which different types of households (and their waste) are distinguished, see Chapter 2.

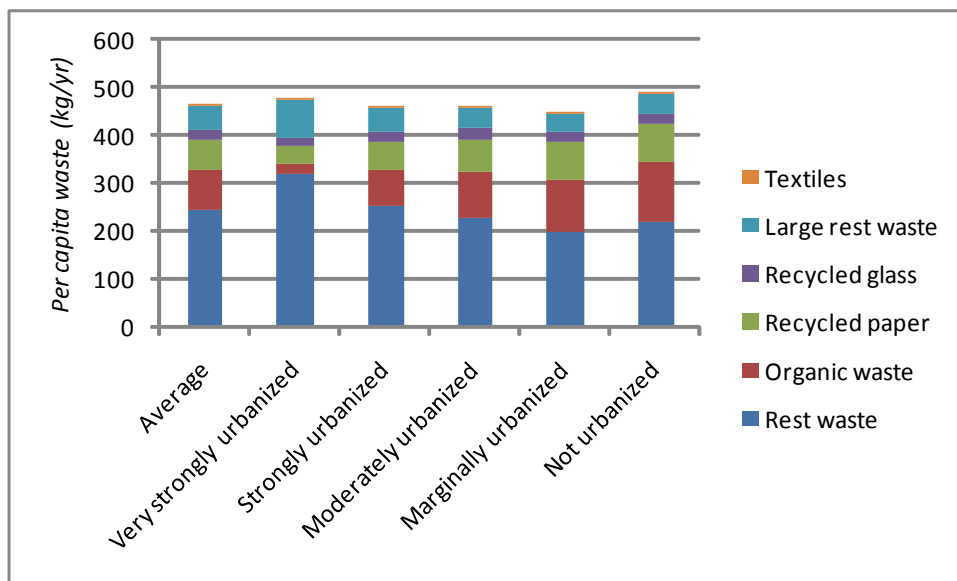


Figure 4.3 Waste production per urbanized areas in the Netherlands, 2006

¹⁷ It should be noted that in the Netherlands municipalities are autonomous to decide whether or not to apply DVR and if so, to select the specific type. The position of the Dutch government is that municipalities are the most appropriate level of governance to take into account specific local circumstances in these decisions. A general obligation to apply DVR nationwide is therefore not being considered.

Increased penetration of DVR systems in the Netherlands

The household types are characterized by household size, household income, and the type of dwelling (houses and apartment buildings), see Table 4.7 in the Appendix to the present chapter.

Despite the fact that DVR has not yet been implemented in larger cities in the Netherlands, the WAP II model has the capability of predicting how successful DVR will be in these areas. Based on an econometric analysis done by Dijkgraaf and Gradus (2004) we can determine the likelihood that a household with certain demographic characteristics (i.e. income, ethnicity, household size, and age) will increase waste separation and recycling as a result of the introduction of DVR. Taking into account the distribution of household types in a municipality we can then determine how waste flows will change after the introduction of DVR. Note that this approach assumes that households, with similar ethnicity, income, size, age, and residence type will respond in the same manner, regardless of whether they live in a village or a city.

With WAP II, we can estimate the changes caused by implementation of DVR for four types of household waste flows: rest waste, recycling paper, organic waste and recycling glass. We run WAP II for municipalities with different waste recovery rates, which are predicted by a linear econometric regression, see Bartelings *et al.* (2005). The WAP II model follows the principle that waste recovery rates decline with increasing levels of urbanisation. The model does not take into account different DVR schemes but assumes that all municipalities introduce a weight-based system. As discussed in the literature section, both the weight-based and expensive bag systems proved to be equally successful schemes¹⁸, while volume-based systems were judged to be less successful. In other words, if a municipality chooses for a volume-based system, the results will be less positive than suggested by the WAPII model.

For the scenarios, we used a stepwise increment of DVR, moving from the current situation in 2006 (CURRENT in Table 4.3) to a Scenario 4 in which almost 80% of the municipalities introduce DVR, representing 56 % of the inhabitants of the Netherlands. For the moderately and (very) strongly urbanized municipalities, the implementation degree does not exceed 50%, because the implementation of DVR becomes disproportionately more difficult in these municipalities.

Table 4.3 Scenarios for implementing DVR (share of inhabitants under DVR scheme)

Urbanization degree	SCENARIO				
	Current	1	2	3	4
Very strongly urbanized	0.0	0.0	0.0	0.0	25.0
Strongly urbanized	8.8	8.8	8.8	8.8	25.0
Moderately urbanized	20.2	20.2	20.2	50.0	50.0
Marginally urbanized	44.4	44.4	100.0	100.0	100.0
Not urbanized	36.4	100.0	100.0	100.0	100.0
Municipalities with DVR	31.7	51.1	70.5	76.3	79.0
Inhabitants with DVR	20.8	28.9	40.8	47.2	56.0

¹⁸ Note that weight based systems may be particularly difficult to implement in extremely urbanized areas, for which a system of 'expensive bags' will be more appropriate.

In WAP II Model, the collection of recycling paper and glass is for free, because these waste flows are not collected at the curb but with community containers. For the collection of waste, household do pay a fee. In a number of municipalities, organic waste is not separately collected from Rest waste. Households pay a fee for waste collection, which includes the costs for composting or incinerating as well as transportation costs. The fee for the collection of organic waste is not as high as the fee for the collection of Rest waste. The costs of composting excluding transportation costs (€35 per tonne) are lower than the costs of incinerating (€100 per tonne).¹⁹ In the case of DVR systems, these differences in costs are incorporated in the fees for waste collection.

Apart from recycling it maybe that increasing implementation of DVR also stimulates waste prevention. To simulate this in the WAP II model, 2 types of consumption goods are taken into account: waste intensive goods and waste extensive goods. However, changes in the costs of waste collection hardly have any impact on consumption patterns. Figure 4.4 shows that the total amount of waste will decline by 3 percent when 80 percent of the municipalities implement DVR (see Scenario 4). The main reason for this minimal effect is that the expenditures on waste collection are only a minor part of the total expenditures of households. Therefore, the incentive to waste prevention is small.

Although waste production only declines by 3 percent when 80 percent of the municipalities implement DVR (i.e. Scenario 4), Figure 4.4 shows that the changes to the different waste streams generated is large. The total amount of rest waste will decrease by 11.6 percent and the organic waste by 5 percent. The waste flows of recycled glass and paper will increase by almost 11 percent and 21 percent, respectively.

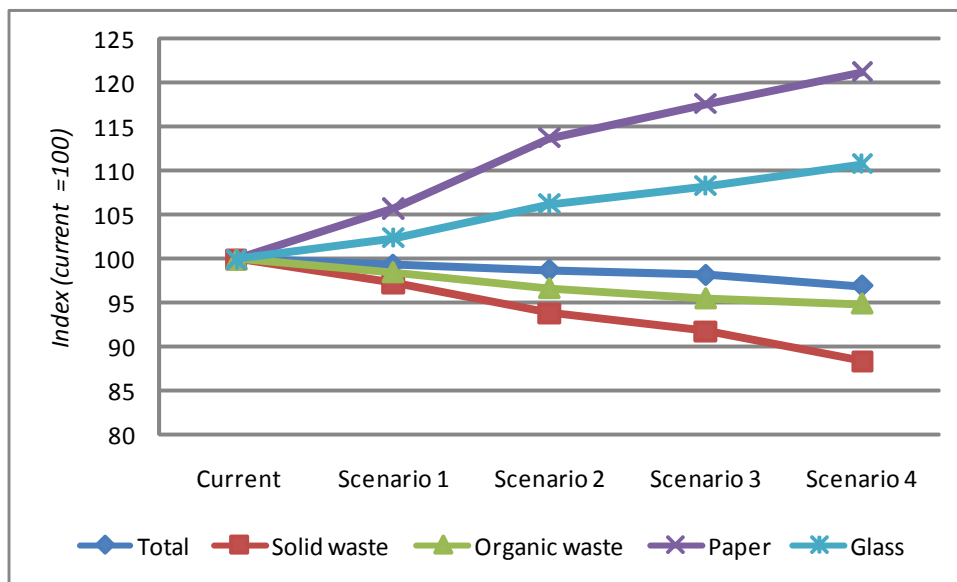


Figure 4.4 Development of household waste flows when increasing DVR implementation (positive fee for organic waste collection)

¹⁹ These costs are derived from www.senternovem.nl. The transportation costs are approximately 30 per tonne.

Based on the results of WAP II, we estimate the environmental impacts of the changes in household waste flows. For these estimations, we use the environmental themes intensities as reported in CE (2008) for waste flows. A linear relationship between the volume of the waste flow and its environmental impact is assumed. The intensity differs across the environmental themes.

In Scenario 4, the total amount of rest waste declines with 20% which has a negative impact on the climate change indicator, see Table 4.6 in the Appendix. The reason for this negative impact is that less waste is used waste incineration (WIT) plants, so that these WITs require more primary energy sources to produce the same amount of total energy (5.5%). Environmental themes like human toxicity, soil eco-toxicity and smog formation improve when more municipalities would implement DVR.

Collection of organic waste for free

In the above scenarios, the fee for collecting organic waste is lower than the fee for collecting rest waste. Yet, the collection fee for organic waste is not equal to zero. If the collection of organic waste would indeed be for free, as represented in the CURRENT' scenario, the present situation would change as well. As shown in Figure 4.5 the total amount of organic waste would be 14.5% higher in the CURRENT' scenario compared to CURRENT. The amount of recycling paper declines with 7.5%, which has a significant impact on the environmental indicators of the CURRENT' scenario compared to the CURRENT scenario. The environmental indicator would change as well in CURRENT' compared to CURRENT, see Table 4.6. The ozone layer and human toxicity indicators would be 4% and 7% higher. Primary energy use and the climate change indicator are 4% lower in CURRENT', because less primary energy is necessary for paper recycling. Paper recycling requires a significant amount of primary energy input according to CE (2008). The scenarios with increasing DVR implementation will be compared to the CURRENT' scenario.

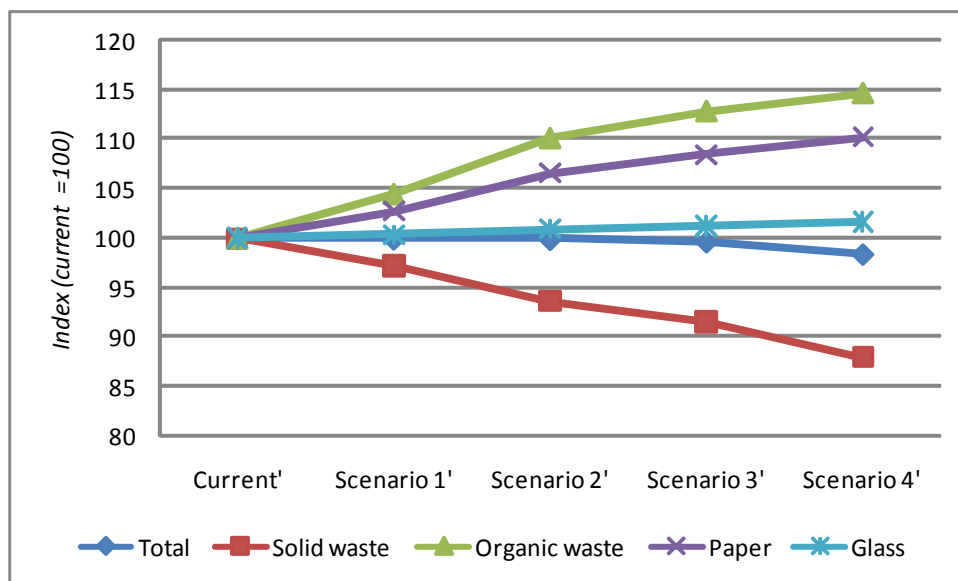


Figure 4.5 Development of household waste flows when increasing DVR implementation (zero fee for organic waste collection)

For the WAP II model including a zero fee for the collection of organic waste (CURRENT'), we ran similar scenarios as the model with a positive fee, see Table 4.5 in the Appendix. In these scenarios, households only pay for the collection of rest waste. Since the rest waste flows decline in all alternative scenarios, the costs of waste collection decline as well, while the consumption slightly increase (less than 0.1%).

If the collection of organic waste would be for free, the total amount of waste (in SCENARIO 4') would decline by 1.6% (compared to CURRENT'). The main reason for this is that the amount of organic waste increases by 14 percent, while it declines in the case of a positive fee for collection (see Figure 4.3). In addition, the amount of rest waste declines more when the collection of organic waste is for free (12% in SCENARIO 4') than when it is not (11.6% in SCENARIO 4).

The disadvantage of the collection of organic waste for free or for a lower price than rest waste is that the quality of this waste will decline (see Bartelings 2003, Bartelings *et al.* 2005). Then either the output of the composting process has lower quality or extra efforts have to be taken to guarantee the quality. In the worst case scenario the organic waste cannot be composted anymore and needs to be incinerated instead.

As for the environmental impacts, the climate change indicator increases by 1 percent in SCENARIO 4' (compared to CURRENT'), while the primary energy use declines by 1 percent. Environmental themes like human toxicity, soil eco-toxicity and smog formation improve when more municipalities would implement DVR, which is a similar result as the results for a positive fee for the collection of organic waste.

Effectiveness of different types of DVR

With the WAP II model, we cannot distinguish between different DVR systems. Although the total amount of waste and the amount of rest waste would decrease due to higher implementation rates of DVR in municipalities, the type of DVR is important to consider due to effectiveness and the potential to implement DVR in strongly urbanized areas. According to Kinnaman and Fullerton (1996), the weight-based pricing systems are more effective than volume-based DVR systems. Dijkgraaf and Gradus (2004) confirm this result, but they add that the 'pay per bag' system is as effective as weight-based systems. Weight-based pricing systems are more expensive systems than 'pay per bag' systems and more difficult to implement in strongly urbanized areas due to the high share of apartment buildings. Curbside collection with individual containers is not applicable in urbanized areas. In South Korea, a 'pay per bag' system is applied nationwide including in a highly densely populated city like Seoul.

4.6 Conclusions

Waste collection charging at differential and variable rates (DVR) has shown to be an effective instrument in reducing the supply of unsorted household waste and to promote recycling. An obvious precondition for a high level of effectiveness is the presence of adequate infrastructure for separate collection. Moreover, many of the often cited objections against DVR appear not to be valid: (1) it can be successfully applied in large cities (as has been shown in some Dutch cities as well as abroad), (2) it does not lead to large increases in fly-tipping and littering, and (3) it does not increase the overall cost of waste collection for municipalities.

Compared to the present situation in the Netherlands (in which 20% of the population is subject to a DVR system), the total amount of household waste would slightly decline if DVR is applied more widely and the separation of waste would significantly increase. Note that it does matter whether the collection of rest waste flows is for free. The environmental impacts are mixed. Environmental themes such as human toxicity, soil eco-toxicity and smog formation show improvements. The use of primary energy resources increases as well as the climate change indicator, because the amount of household rest waste declines and is substituted by primary energy sources.

Despite the strong evidence of the potential positive role of DVR in improving waste management, it remains difficult to promote DVR at a national level. According to the Environmental Protection Act in the Netherlands, municipalities are responsible for the collection of waste. This decentralised approach meets with the idea that local decision makers are better capable of designing a tailor-made system taking into account local conditions. The actual choice of implementing DVR is up to municipalities.

Appendix to Chapter 4

Table 4.4 Development in household waste flows (Mton) with the increasing implementation of DVR

	Total*	Rest waste	Organic waste	Recycling paper	Recycling glass
<i>Positive fee for the collection of organic waste</i>					
CURRENT	6,639	3,738	1,324	1,212	366
SCENARIO 1	6,599	3,639	1,304	1,282	374
SCENARIO 2	6,558	3,511	1,279	1,380	389
SCENARIO 3	6,519	3,433	1,264	1,426	396
SCENARIO 4	6,437	3,306	1,256	1,470	405
<i>Zero fee for the collection of organic waste</i>					
CURRENT'	6,710	3,728	1,516	1,121	345
SCENARIO 1'	6,707	3,625	1,584	1,152	346
SCENARIO 2'	6,705	3,492	1,669	1,195	348
SCENARIO 3'	6,687	3,411	1,711	1,216	349
SCENARIO 4'	6,604	3,279	1,739	1,235	351

* Note that the total amount of waste only represents the total sum of the four waste flows considered.

Table 4.5 Development of environmental themes per DVR scenario compared to the CURRENT scenario

	SCENARIO			
	1	2	3	4
Base=100	CURRENT	CURRENT	CURRENT	CURRENT
Total amount of waste	99.40	98.78	98.19	96.95
<i>Environmental theme</i>				
Abiotic exhaustion (ADP)	91.42	79.92	73.77	65.79
Climate change (GWP100)	101.83	104.52	105.57	105.93
Ozone layer (ODP)	91.65	80.52	74.61	67.03
Human toxicity (HTP)	91.24	78.45	70.89	58.83
Smog formation (POCP)	91.54	80.69	75.21	68.75
Energy use (MJprim)	101.67	104.19	105.19	105.50

Table 4.6 Development of environmental themes per DVR scenario compared to the CURRENT scenario (zero fee for organic waste collection)

	CURRENT'	SCEN. 1'	SCEN. 2'	SCEN. 3'	SCEN. 4'
Base=100	CURRENT	CURRENT'	CURRENT'	CURRENT'	CURRENT'
Total amount of waste	101.06	99.96	99.93	99.67	98.42
<i>Environmental theme</i>					
Abiotic exhaustion (ADP)	104.77	93.82	85.57	80.95	74.31
Climate change (GWP100)	96.96	100.58	101.46	101.68	101.15
Ozone layer (ODP)	104.45	93.93	85.83	81.33	74.90
Human toxicity (HTP)	107.14	94.05	86.14	81.22	72.73
Smog formation (POCP)	100.24	91.98	81.36	75.74	68.75
Energy use (MJprim)	95.94	99.97	100.08	99.92	99.05

Table 4.7 Distribution of household types (high/low building, household size, income class) per category of urbanization rate

Building	Household size	Income class	Very strongly urbanized	Strongly urbanized	Moderately urbanized	Marginally urbanized	Not urbanized
High	Single person	Low	8.4	3.8	2.6	2.1	1.8
High	Two persons	Low	2.0	1.0	0.8	0.6	0.5
High	Multiple persons	Low	6.3	5.6	5.4	5.3	4.9
High	Single person	Middle	3.9	1.9	1.3	1.1	1.1
High	Two persons	Middle	0.9	0.5	0.4	0.3	0.3
High	Multiple persons	Middle	2.9	2.7	2.8	2.8	3.0
High	Single person	High	5.6	3.4	2.6	2.2	1.7
High	Two persons	High	1.3	0.9	0.8	0.7	0.5
High	Multiple persons	High	4.2	5.0	5.5	5.5	4.8
Low	Single person	Low	15.3	11.6	9.1	7.9	7.7
Low	Two persons	Low	3.6	3.1	2.7	2.4	2.3
Low	Multiple persons	Low	11.5	17.1	19.3	20.4	21.4
Low	Single person	Middle	7.0	5.6	4.6	4.2	4.6
Low	Two persons	Middle	1.6	1.5	1.3	1.3	1.4
Low	Multiple persons	Middle	5.3	8.3	9.8	10.9	13.0
Low	Single person	High	10.2	10.2	9.2	8.3	7.5
Low	Two persons	High	2.4	2.7	2.7	2.5	2.3
Low	Multiple persons	High	7.6	15.1	19.3	21.4	21.1

* Urbanization rate is the number of addresses per kilometre squared.

Note that the share of single person households is high in very strongly urbanized municipalities. Multi person households have high shares in marginally and not urbanized municipalities. In marginally and moderately urbanized areas, low buildings have high shares.

5. Taxes on raw materials and products

5.1 Introduction

Instead of imposing a tax or charge on waste, i.e. at the end of the waste chain, as discussed in the preceding chapters, it can also be done at the beginning of the chain. The taxation of (non-renewable) raw materials provides an incentive to economize on materials and resources at the very beginning of a product's lifecycle, and may make the use of secondary materials more attractive. Taxes and charges aiming at waste prevention and reduction can also be imposed somewhere along the chain. A product tax may be an appropriate option if it is not feasible to give effective price incentives in the waste stage, or if there are other (not waste related) product-specific environmental problems that call for discouraging the product's use. Product charges, on the other hand, may be introduced to finance the separate collection, processing and recycling of waste streams from the products on which they are levied.

5.2 Theory and literature

5.2.1 Taxes on raw materials

Economists generally argue that policy instruments should be directly addressed at market failures. If the market failure exists at the end of the chain (in the waste stage), imposing a tax at the beginning of the chain (raw materials) would therefore not be appropriate, unless there are specific market imperfections (such as high transaction costs) along the chain that would justify it. For example, Dinan (1993) shows that taxing virgin materials would be an inefficient method of reducing waste disposal. Söderholm (2006) argues that the case for taxing natural resources for conservation reasons may not be particularly strong, as it discourages investments in the resource base (e.g. afforestation). Moreover, a tax on virgin material can only correct for external costs resulting from extracting or harvesting virgin materials, but not for external costs resulting from waste disposal. The reason for this is that at the 'cradle' stage it is obviously unknown how the material will eventually be disposed of, i.e. what its 'grave' will be like.

On the other hand, there might be a case for raw materials taxes in the context of more general policy objectives such as promoting recycling, reducing material intensity in production and consumption, or discouraging the use of specific materials in particular. Leaving aside the question if such objectives would be optimal from a social welfare point of view, taxing raw materials might well be a cost-effective instrument to achieve them, as it improves the competitive position of the alternatives.

Concerning the types of raw materials that might be suitable for taxation, one might consider that the relatively high weight of construction materials such as sand and gravel makes it uneconomic to transport them over long distances. This reduces the likelihood that taxing them would shift the production abroad, to countries without such taxes.

5.2.2 Taxes and charges on products

The impact of a product tax will largely depend on its specific design. In order to be an effective environmental policy instrument, it will need to address the environmentally undesirable properties of the product as closely as possible. For example, a general tax on packaging may be justified if the primary objective is to reduce resource use and waste in general, but it may need to be differentiated or confined to specific materials if it is intended, for instance, to reduce the amount of non-biodegradable waste. The specific circumstances (including market conditions) may also call for a product tax to be embedded in a package of instruments, including non-economic ones. Eichner (2005), for example, concludes that a tax on consumption goods, in combination with a relative recycling standard, a material subsidy and a subsidy on recycling services, might be an appropriate policy instrument under conditions of imperfect competition in the recycling industry. Likewise, Dinan (1993) found that a combination of a disposal tax and a reuse subsidy would be an efficient method of reducing waste.

Introducing a new tax scheme on specific products can be costly in terms of administrative, management, implementation and enforcement costs. This is especially true if a large number of relatively small actors is involved. The instrument is therefore mainly suitable for products that are consumed in large numbers and/or with a small number of producers and importers. Alternatively, differentiations in existing product taxes (such as VAT and car taxes) can be applied, but this option has some obvious limitations (see also chapter 7).

5.3 The Dutch situation

Apart from taxes on the extraction of energy and groundwater, there are currently no national taxes on natural resources and raw materials in the Netherlands. Provinces, however, can levy a charge ('ontgrondingsheffing') on the extraction of soil materials (sand, gravel, clay, etc.). This charge is only intended to cover (part of) certain costs that are directly related to the extraction, e.g. the compensation for damage caused by the quarry. It is therefore not intended as an incentive to economize on primary raw materials, and due to the low level of the charge rates²⁰ it is also unlikely to do so.

In the Netherlands, there is only one specific product tax that is directly related to waste: the packaging tax. In addition, several (explicit or implicit) product charges exist, which are levied to enable producers and importers to fulfil the waste management duties related to their 'producer responsibility'.

5.3.1 The packaging tax

Description

The packaging tax, based on article 81 of the Environmental Taxes Act, entered into force on 1 January 2008. The tax applies to packaged products as well as to packaging as such. The point of interference is at the beginning of the value chain in the Netherlands. In other words, the tax has to be paid by the importer or producer who puts the

²⁰ Usually below €0.10 per m³; this is in the order of magnitude of 1% of the market price.

packaging at someone else's disposal for the first time in the Netherlands. This means that, for instance, packaging that has a deposit and other packaging for multiple use are taxed only once. The tax has to be paid also by importers who remove and dispose of the packaging of their imported products directly after these have entered the Netherlands.

There is a threshold of 15,000 kilograms of packaging produced or imported per year; up to that amount no tax has to be paid. This threshold effectively restricts the number of firms liable to pay the tax to some 8,000 to 10,000. Together these account for 95% of the packaging brought on the Dutch market.²¹

Table 5.1 shows the packaging tax rates as they apply in 2008.

Table 5.1 Packaging tax rates, 2008 (€ per kg)

Material	Primary packaging	Secondary & tertiary packaging
Glass	0.0456	0.0160
Aluminum and aluminum alloys	0.5731	0.2011
Other metals	0.1126	0.0395
Plastics	0.3554	0.1247
Bioplastics	0.1777	0.0624
Paper and cardboard	0.0641	0.0225
Wood	0.0228	0.0080
Other materials	0.1017	0.0357

Effects and effectiveness

The packaging tax has a hybrid character. On the one hand, it is aimed at generating revenues, both for the government's general budget and for specific waste related activities (Waste Fund; see Chapter 7). On the other hand, it is also intended to contribute to a reduction of the amount of packaging waste, to encourage reuse, and to stimulate the use of packaging materials that are less environmentally harmful. In other words, it has both a financing and an incentive function. To what extent this incentive function is actually achieved cannot yet be determined in this stage. Any impacts are difficult to measure, given the recent introduction of the tax. Its expected effectiveness in terms of reducing the amount of packaging should therefore be based on foreign experiences with a longer history (see section 5.4).

Expectations and prospects

From 2009 on a structural revenue of €365 million is envisaged. In order to reduce the impact of the tax on inflation, the rates for 2008 have been set at a lower level and the revenue in this year will be only €240 million.

In July 2008 the state secretary of Finance announced that the packaging tax will be simplified in some respects. Among others, the distinction between primary and secondary/tertiary packaging will be abolished as of 1 January 2009.²²

²¹ Source: Belastingplan 2008.

²² Source: News release Ministry of Finance, 18 July 2008.

5.3.2 Producer responsibility and waste management fees

Description

For certain product categories (batteries, electric and electronic appliances, and packaging) producer responsibility has been introduced by means of regulation (based upon article 10.15 of the Environmental Protection Act).²³ This implies that producers and importers have a legal (co-)responsibility for the management of these products in their waste stage, including its financing. For a number of product groups the financing takes place by means of waste management fees (also called disposal fees). Producers and importers can also voluntarily accept producer responsibility and ask the Minister of Environment to make their waste management fees generally binding (based upon article 15.36 of the Environmental Protection Act). This has happened, for instance, for paper and cardboard.

Table 5.2 shows the rates of the waste management fees in 2008 for electr(on)ic equipment.

Table 5.2 Rates of the waste management fees for electr(on)ic equipment. 2008

Products	Waste management fee (€)
Fridges and freezers	17
TV sets	8
Washing machines, dryers, dishwashers, cookers, ovens	5
DVD recorders and players; video-DVD combinations	3
Coffee machines, vacuum cleaners, deep fryers	1

Source: Stichting NVMP.

Producers and importers of other product groups, such as batteries and ICT equipment, have refrained from charging a waste management fee that is 'visible' to the consumer. They do, however, pay a fee to the organisation that takes care of the waste management on their behalf. An example is the Batteries Foundation (Stibat), which manages waste batteries and charges a fee with a rate depending on weight, ranging between EUR 0.02 and 0.54 per battery (for button cells EUR 0.003).²⁴

Effects and effectiveness

Producer responsibility implies that the consumer can hand in certain discarded products at no cost at the retailer or municipality for recycling or disposal. For example, through the collective 'NVMP'²⁵ scheme more than 70,000 tonnes of discarded electr(on)ic appliances per year are collected²⁶, through 'ICT-Milieu' 20,000 tonnes of ICT equipment²⁷, and through 'Stibat' almost 3,000 tonnes of batteries²⁸. However, since there are no positive incentives to keep these products separated from other waste

²³ In addition, producer responsibility exists for cars and tyres, but these products do not belong to the types of waste that the current report addresses.

²⁴ Source: Stibat, Beheerplan 2003-2008 (included in Annual Report 2006).

²⁵ Stichting Nederlandse Verwijdering Metalelektro Producten.

²⁶ Source: NVMP.

²⁷ Source: ICT-Milieu.

²⁸ Source: Stibat, Jaarverslag 2006.

streams, a considerable part still ends up in the general household waste. For example, of the 18.5 kilogrammes of electr(on)ic equipment discarded per inhabitant per year in the Netherlands only 5.7 kg (slightly above 30%) makes it into the 'NVMP' and 'ICT-Milieu' schemes.²⁹

In 2007 the Ministry of the Environment (VROM) presented a study on the side effects of producer responsibility.³⁰ This revealed, among others, that an efficient and high-quality system of collection and recycling is in place. Improvements are possible in terms of financial transparency of the implementing organisations. In some cases (too) large funds have been created, due to rates that were set at a too high level and/or due to lower than expected costs. Furthermore, the collective arrangement of producer responsibility does not provide incentives for 'design for recycling'.

Expectations and prospects

The system of 'visible' waste management fees for electr(on)ic equipment will be terminated by 2011 (for large household appliances by 2013). From then on, the costs of waste management will be incorporated in the product price without separate specification, as is already the case for other product groups.

According to the Minister of VROM, funds at the producer organisations should not exceed the amount of one year's turnover. On this subject, and on the issue of financial transparency, agreements with the organisations will be made.³¹

5.4 Foreign experiences³²

5.4.1 Taxes on raw materials

Several EU countries apply taxes on the extraction of natural resources. Gravel and sand are the most commonly taxed materials. Tax rates vary widely, but usually lie between EUR 0.10 and 1.00 per m³.

A study by the EEA (2008) assessed the impact of taxes and charges on sand, gravel and rock in four EU Member States: the Czech Republic, Italy, Sweden and the UK. It clearly shows that countries with relatively high tax/charge rates also have relatively high recycling rates (see Table 5.3). However, the study emphasizes that in all cases the tax/charge is just one factor among many, including other policies, technical factors, and specific national circumstances.

Generally, taxes on raw materials do not seem to have much impact in terms of a reduction in the demand for virgin materials and a shift towards recycled materials. This is confirmed by the experience in Denmark, where the waste tax has played a much larger role in stimulating the recycling of construction materials than the tax on raw materials (Söderholm, 2006).

²⁹ Source: www.witteveenbos.nl/nieuws/bericht-509.html (24.07.2008).

³⁰ Second Chamber of Parliament, 2006-2007, 27664, nr. 48.

³¹ Second Chamber of Parliament, 2007-2008, 27664, nr. 52.

³² For details by country, see Appendix I.

Table 5.3 Taxes and charges on sand, gravel and rock and recycling rates in four EU countries

	Czech Rep.	Italy*	Sweden**	UK
Tax rate in % of average price	3	5	12	20
Recycling rate	5	1	11	25

* Two regions: Lombardy and Emilia-Romagna.

** The Swedish tax only applies to gravel.

Source: EEA, 2008

Taxes on aggregates may have some unintended side effects. For example, in Sweden the tax on gravel may have led to a substitution by crushed rock, which requires approximately three times more energy per tonne of material. In the UK, a shift of trade in aggregates took place from Northern Ireland (with aggregates tax) to Ireland (without tax) (EEA, 2008).

5.4.2 Taxes and charges on products

Various countries apply product taxes to products that generate waste (see Appendix I). The most widely taxed product categories include: batteries; packaging and (plastic) carrier bags; miscellaneous disposable products, such as tableware; fluorescent lamps containing mercury; and electr(on)ic appliances.

South Korea is probably the country with the widest range of waste related product charges, which cover, among others, products such as chewing gum, diapers, and plastic construction materials and furniture.

It is not always clear if the product tax is intended as an incentive for waste reduction (by reducing demand for the taxed product and stimulate the sales of alternatives that are less waste-intensive). It seems that in most cases the primary function is a revenue-raising one, either for the general budget or to raise funds for the management of the waste from the taxed product. Only in a few cases (such as the Irish tax on plastic bags) a clear incentive function is explicitly stated as well as empirically observed in practice.

5.5 Case study

We have seen that, under certain conditions, taxes on raw materials and products can be useful components of a waste policy package. Generally speaking, they should relate to goods that are traded in relatively large volumes and a limited number of taxpayers should be involved, so as to minimize the costs of administration and enforcement.

Taking into account the priority waste streams in Dutch waste policy, the following raw materials and products have been selected as candidates for further analysis:³³

- Sand and gravel;
- Paper and cardboard;
- Primary aluminium.

³³ Primary PVC is also one of the priority waste streams in the Dutch waste policy. However, the analysis of PVC is hampered by the fact that no specific data are available on its production, consumption and waste flows. Therefore, it was decided to exclude this material from the analysis. More information on PVC in the Netherlands is found in the Appendix.

In addition, the option of increasing the tax rates of the existing tax on packaging materials will be analysed.

An attempt is made to analyse the impact of introducing a tax on the above mentioned raw materials. The aim of these taxes is to encourage recycling of these priority waste streams. Note that the analysis addresses sand and gravel in a qualitative manner and paper and aluminium in a quantitative manner.

In addition, the option of increasing the tax rates of the existing tax on packaging materials will be analysed.

5.5.1 Sand and gravel

Gravel extraction in the Netherlands has decreased from 16 Mtonne in 1980 to 3.3 Mtonne in 2006. The imports of gravel have increased. Since the 'Use' rows in Table 5.4 are only used for the balancing of the production and consumption, the use of gravel is underestimated. Note that the use of secondary gravel in production is not included, because it is not systematically registered. CE (2004) has calculated the amount of secondary gravel in the production of gravel to be 4 Mtonne in 2001. This is about 13% of the total gravel production in 2001.

Table 5.4 Development of Sand and gravel extraction and use in the Netherlands, 1980-2006

	1980	1990	2000	2003	2004	2005	2006
<i>Gravel (Mton)</i>							
Extraction	16,0	8,8	6,5	4,5	3,8	3,6	3,3
Import	12,1	14,1	10,8	20,6	17,7	16,3	16,2
Export	4,0	0,9	2,2	1,2	0,7	0,3	0,3
Use	24,1	22	19,4	19,3	15,5	18,5	17,3
Total	28,1	22,9	21,6	20,5	16,2	18,8	17,6
<i>Sand for concrete production (Mton)</i>							
Extraction	21,3	21,8	21,2	15,2	13,6	14,0	14,7
Import	6,9	7,9	11,0	17,1	13,0	9,8	12,4
Export	8,9	8,4	8,7	6,5	4,4	2,5	4,6
Use	19,3	21,3	22,5	22,7	22,6	22,7	21,4
Total	28,2	29,7	31,2	29,2	27,0	25,2	26,0
<i>Sand (million cubic meters)</i>							
Extraction	53	48,3	61,5	46,4	42,6	45,5	47
Import							
Export	2,0	2,0	2,5	3,3	3,2	4,2	4,5
Use	51,0	46,2	58,0	43,1	39,5	41,3	42,6
Total	53,0	48,2	60,5	46,4	42,7	45,5	47,1

Note that Extraction + Imports = Export + Use = Total. Secondary use is excluded.

Source: MNC (2008) and Koopmans (2006).

Sand extraction in the Netherlands has declined as well in the period 1980-2006, although the decreases are rather small. For sand, there is no information on the use of secondary materials available.

A possible tax on aggregates (sand, gravel and clay) was already studied some years ago (CE, 2000), with a view to introduction in 2001. The tax rate being considered was

NLG 1.75 (almost EUR 0.80) per tonne, with an exemption for sand from the North Sea. The study concluded that this tax would reduce the amount of aggregates extracted on-shore by 10 to 17 million tonnes in 2005, whereas the offshore mining of sand in the North Sea would increase by some 7 million tonnes. There would be hardly any impact in terms of a reduction in the use of raw materials (at least in the short term) and in terms of a shift to secondary raw materials (even under a scenario with a much higher tax rate of NLG 6 or EUR 2.70 per tonne). The proposed tax would lead to a reduction in economic activity, mainly in regions close to the Belgian and German border.

The main reason why a tax on aggregates has not (yet) been introduced is the fact that it appeared not to be possible to use the revenues from the tax to fully compensate the affected industries.³⁴ However, since 2003 there has also been a change in the general vision on the issue of resource extraction and the government's role in this issue. The Dutch government has decided to withdraw itself from the area of 'ensuring sufficient supply of building material resources' and leave this to market forces. Moreover, the policy is now aimed at stimulating the extraction of these resources in the Netherlands in a socially acceptable way, so as not to pass the spatial problems involved onto other countries.³⁵ Obviously, a tax on the extraction of raw materials would not fit anymore in such a policy, unless neighbouring countries would take similar measures.

On the other hand, the Dutch policy is also still aimed at a more efficient use of raw materials and an increase of their recycling. Against this background, a tax on primary aggregates could still make sense if it is levied not only on their extraction, but also on imported materials (as is the case in the UK and Denmark). The point of interference in the chain would then be the firm that puts the product on the Dutch market for the first time.

As noted above, foreign experiences show that the impact of taxes on raw materials in terms of demand reduction and a shift towards recycled materials is generally very limited, even if it is levied both on domestic and imported materials. However, the scarce evidence available does not allow us to make any quantitative estimates. Given the fact that 97% of the construction and demolition waste in the Netherlands already finds its way to a useful application³⁶, the additional 'gains' that can be achieved in this respect will be limited anyway.

A tax on the extraction of sand, clay and gravel could replace the presently existing provincial charge ('ontgrondingenheffing'; see Section 5.3). Provinces would probably need to be compensated for this loss of own revenue.

5.5.2 Paper, aluminum and packaging

Paper and cardboard

Households and service sectors are large users of paper and cardboard. Table 5.5 shows the development in new paper and collected paper for recycling in the Netherlands. In

³⁴ See Belastingplan 2002.

³⁵ See, for instance, a letter by State Secretary Huizinga to the Second Chamber of Parliament, 3 July 2008 (31200 XII 93).

³⁶ Source: Milieu- en natuurcompendium.

2007, 3,462 Ktonne paper products were sold in the market, while 2,730 Ktonne paper was recovered for recycling purposes. This implies that the recovery rates of waste paper are in the range of 75-80%. The remaining gap between paper products sold and recovered waste paper includes paper still stored at companies or households and waste paper that is not separated but mixed with disposable waste. In 2005, 2.7 Mtonne waste paper was collected, while 2.5 Mtonne was actually used for recycling. A small portion of the collected paper was not suitable for recycling due to its quality.

Table 5.5 New and recovered paper for household and Service sectors, 2005-2007

Variable	2005	2006	2007
New paper on the market in Ktonne (PRN)	3,270	3,380	3,462
Recovered paper in Ktonne (PRN/FNOI)	2,510	2,600	2,730
Recovery rate (compared to new paper sold)	76.8%	76.9%	78.9%
Collected waste paper in Ktonne (SenterNovem)	2,703	--	--
Collected waste paper at households in Ktonne (FNOI)	1,130	1,175	--
Share of household in collected paper	41.8%	--	--

Source: PRN (the Netherlands Association of Paper Recycling) and the FNOI (Federation for the Dutch Recovered Paper Industry) and SenterNovem (2008a).

From SenterNovem (2008a), we know that 2% of the collected waste paper is either landfilled or incinerated. The rest of the collected waste paper is separately collected for reuse or recycling.

Aluminum

According to the Association of Aluminum Producers in the Netherlands (i.e. Aluminium Centrum), the total annual use of aluminum in the Netherlands is 30 kilograms per person. This means that annually approximately 495 Ktonnes of aluminum is produced (assuming that the population is 16.5 million people). Aluminum is primarily used in the construction and transport sector but also for packaging. With respect to recycling, The Dutch Aluminum Centre reports an overall recycling rate of aluminum of 72% in 2000 (more recent numbers are not available). The recycling rates in the construction and transport sector are even 94%. For other waste flows, like household waste, the recycling rate will be much lower than 72%, although exact recycling rates are not available. From household waste we know that 2 Ktonnes of metal packaging including aluminum are collected (SenterNovem 2008a).

One of the main advantages of aluminium recycling is the large savings on energy use during the production stage. Using recycled aluminium in the production stage requires only 5% of the energy input of using raw material to produce aluminium. The recycling of 1 kilogram of aluminium saves approximately 14 kWh energy, which implies that recycling of 1 kg of aluminium requires less than 1 kWh energy (Waste-online, 2008).

Packaging

Table 5.6 shows the materials used for packaging of consumption products. Paper and cardboard cover two-third of the total amount of packaging. Plastic and glass have a share of 20% in packaging. Metal (84%) and glass (78%) have the highest recovery rates. Paper and cardboard also have a recovery rate of almost 72%. For plastic, the recovery is 34%. For metal, it is relatively easy to separate it before collection or to

separate it from unsorted waste flows. For glass, people are traditionally used to separate glass from rest waste and community glass containers are widely available (Annual report Commission Packaging 2005).

Table 5.6 Input and recovered materials of packaging in the Netherlands, 2005

	Materials in packaging			Recovery rate %	Share of materials in packaging	
	Input Ktonnes	Recovered Ktonnes	Incinerated Ktonnes		Input %	Recovered %
Paper/cardboard	1,465	1,051	2	72	44	53
Plastic	592	131	72	22	18	7
Glass	545	423		78	16	21
Metal	211	177		84	6	9
Wood	533	206	39	39	16	10
Total	3,346	1,986	113	63	100	100

Source: Annual report Commission Packaging 2005.

For a number of materials such as glass and paper and cardboard, minimum prices for separate collection are applied. At this moment, the minimum price for paper and cardboard recovery is €20 per tonne.

Analysis

To analyse the effects a tax on virgin material (like paper and aluminium) can have on the use of recycled materials and the generation of recyclable waste the WAP I model has been used.

Using a scenario where a virgin material tax is introduced making the price of virgin material twice as expensive, it is calculated how much both households and the service sector will increase separation of recyclable waste. Figure 5.1 shows the change in recycling by both the households and the service sector. It can be expected that the virgin material tax will increase the demand for recycled material. However, to increase the supply of recycled material it is essential that more recyclable waste is collected. The change in recycling is minimal even with the extremely high virgin material tax that is introduced. Households that live in a municipality charging a flat waste fee do not increase recycling at all because they do not experience an incentive to separate more waste. Households that live in a municipality charging a unit-based waste fee increase waste separation slightly (around 1%) they do experience a small price incentive, in the following way: Since the costs of these municipalities for recycling waste is lower, they can decrease the fee for waste collection and thereby provide an incentive to households to separate waste. However, the price incentive for the households is only small, even with a considerable virgin material tax and thus the effects on recycling are minimal.

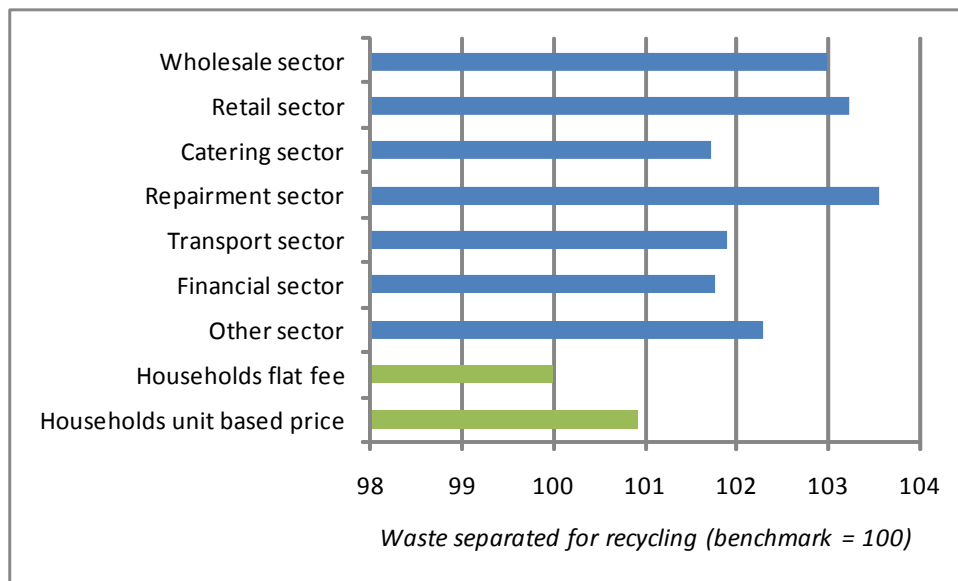
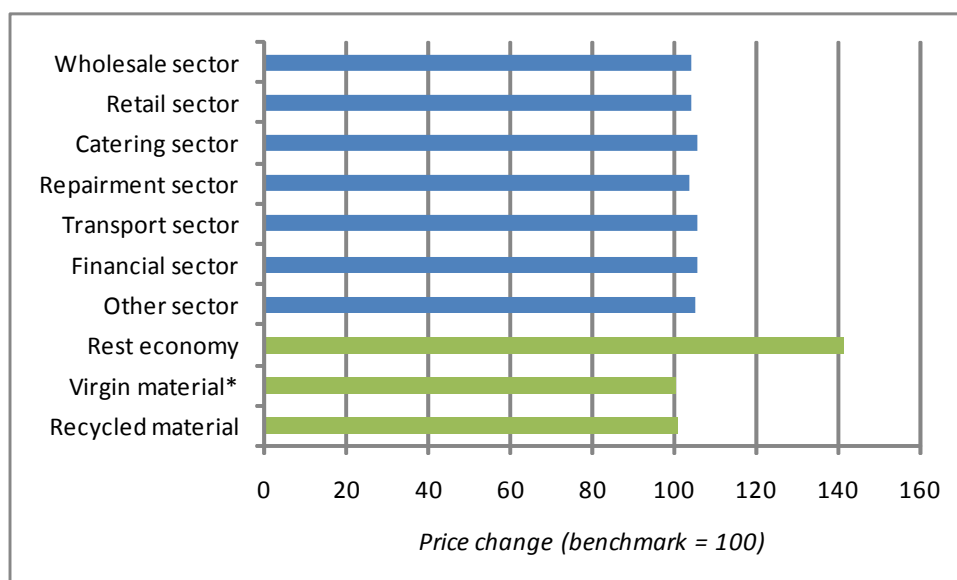


Figure 5.1 Change in waste separation by households and the service sector as a result of an increase in the virgin material tax doubling the price of virgin material

The increase of recycling by the service sector is also minimal. The highest increase is about 3.5% by the repair sector. The substitution elasticities in the model are initiated on data available from the years 1995-2002 (see Bartelings *et al*, 2005). Recycling by the service sector is relatively price inelastic, which the mean reason for the minimal increase in recycling.

The most important reason for these results is that without any accompanying measures the suppliers of recyclable waste, households and the service sector have no incentive to increase waste separation. Therefore, in a closed economy, even if the demand for recycled material increases, the supply cannot increase because the input for recycling (recyclable waste) does not increase in the same proportion. If the suppliers of recyclable waste are not rewarded for increased waste separation, a virgin material tax will fail. A DVR system 'rewards' households for waste separation, however a virgin material tax in the early phases of the product chain provides such a low price incentive in the waste collection/treatment phase of the product chain that even if a DVR system is used, the effects of a virgin material tax on recycling will still be minimal.

The main effect a virgin material tax will have in this case is that it raises the price of those products that use a lot of virgin material. The effects on product prices are shown in Figure 5.2. In the WAP-model virgin material and recycled material are used in the rest of the economy, which is naturally a very substantial sector as it includes all other sectors of our economy. The service sector only uses intermediate deliveries from the rest of the economy. The price of goods produced by the rest of the economy increases substantially as a result of the virgin material tax. Note that the increase in price in the model is quite substantial since import and export of virgin and recycled material is not taken into account. In reality it can be expected that more recycled material will be imported and thus the price increase will not be as large.



* The price of virgin materials is exclusive of the tax increase

Figure 5.2 Change in product prices as a result of an increase in of the virgin material tax making the price of virgin material twice as expensive

Note that the WAP model simulates a scenario under the assumption of a closed economy. Obviously, the recycling market is an international market heavily influenced by world market prices and trade flows. Therefore, introducing a tax on virgin materials may lead to higher imports of recycled materials from other countries. With these international material flows changing, it remains uncertain whether overall more materials are recycled. If no concrete measures are taken to increase separation of recyclable waste either in the Netherlands or in foreign countries, an increase in the demand for recycled material in the Netherlands may very well lead to a higher demand for virgin materials in the countries that we are importing recycled materials from. In other words, in this specific case higher taxes on virgin materials will not increase recycling but only redistribute the use of recycled material and virgin materials internationally.

The virgin material tax would be far more effective if it would be accompanied by measures to increase waste separation by the service sector and the households. Such matters are further discussed in Chapter 4.

Due to the lack of incentives for the producers of recyclable waste to increase recycling, waste streams hardly change as a result of the virgin material tax. Therefore it can also be concluded that the environmental impact of this measure is very limited.

5.6 Conclusions

Taxes on primary raw materials can contribute to a reduction in the use of such materials and to a better competitive position of recycled or renewable alternatives. In order to be effective, they should be part of a coherent policy package and should be designed so as to minimize possible unintended side effects. Likewise, product taxes will need to address the environmentally undesirable properties of the product as closely as possible.

The specific circumstances (including market conditions) may also call for a product tax to be embedded in a package of instruments, including non-economic ones.

In an open economy and under a waste charging system that prevents price signals to be transmitted, any virgin material or product tax should be accompanied by measures to increase waste separation. If the supply of recyclable waste does not increase it will be difficult to switch from virgin/primary materials and products to recycled/secondary materials and products.

6. Deposit-refund schemes

6.1 Introduction

Deposit-refund schemes (DRS) are basically a combination of two instruments: a tax on the purchase of a certain product, and a subsidy on the separate collection of the same product in its after-use stage. Such schemes are widely applied in the area of drinks packaging, mainly on a voluntary basis. The present chapter focuses on the use of deposit-refund systems as an instrument of government policy, i.e. on a mandatory basis.

6.2 Theory and literature

The earliest economic analyses of DRS date back to more than 25 years ago (Bohm, 1981). Stavins (2000) mentions three reasons why DRS may be attractive:

- DRS greatly reduce monitoring costs, as they reduce the incentive to illegal dumping;
- DRS can provide firms with incentives to prevent losses of the material in the industrial process in which it is used;
- Because of inevitable net losses in the production and consumption processes, incentives exist for firms to look for substances that cause less damage to the environment – substances to which the DRS does not apply.

More recently, several authors have studied the conditions under which DRS can be considered a useful policy instrument to attain maximum welfare, and how they should be designed. For example, Aalbers and Vollebergh (2008), using a general equilibrium model, found that DRS can provide the optimal incentives to recycling, landfilling and dumping, taking into account the possibility of waste mixing and the efforts needed to keep waste streams separated. Calcott and Walls (2005) argue that a deposit-refund should be applied to *all* products, combined with a disposal fee (waste tax).

6.3 The Dutch situation

Although the Environmental Protection Act (article 15.32) offers the opportunity to impose mandatory deposit-refund or ‘return premium’ schemes, this instrument is not (yet) used in Dutch waste policy. Voluntary DRS, however, do exist, for example for beer bottles, beer bottle crates, and large PET bottles. These schemes can contribute to reuse and recycling and to a reduction of the amount of packaging waste, as they tend to lead to high return rates (95% for large bottles) (CE, 2001).

The possible introduction of mandatory deposit-refund schemes has been used as a ‘big stick’ to stimulate firms to co-operate in realising the objectives in the area of packaging waste and littering. In this respect, Article 8 of the Decree on the management of packaging, paper and cardboard (waste)³⁷ provides for a deposit-refund scheme for drinks packaging, which will only enter into force if the envisaged targets for collection, reuse or litter reduction will not be realised. Table 6.1 shows the current targets and the actual figures for 2005.

³⁷ Besluit beheer verpakkingen en papier en karton, Staatsblad (Law Gazette) 2005, 183.

Table 6.1 Recycling targets for packaging materials (in %)

Material	Target	Realization 2005
Glass	90	78
Metal packaging	85	84
Plastic packaging	38*	22
Paper and cardboard	75	72
Wooden packaging	25	39
Total for all materials	75	66

* This target is for 2009 and applies to reuse of materials. For 2012 a target of 42% applies.

Sources: Packaging waste management decree, Impulse programme littering; Framework agreement between VROM, industry and Association of Municipalities (VNG) d.d. 27 July 2007; 'Commissie Verpakkingen' (2006).

6.4 Foreign experiences³⁸

Traditionally, deposit-refund schemes for beverage packaging are in place on a voluntary basis. In some countries the use of such schemes has been stimulated by the fact that high return rates allow companies to enjoy privileges, e.g. a reduced rate or exemption from packaging taxes.

A number of countries, particularly in Scandinavia and Central and Eastern Europe, as well as states in the USA and provinces in Canada, have been applying mandatory deposit-refund systems for several years. Germany has recently joined their ranks. The types of beverage containers covered differ (e.g. they may or may not include non-refillable containers such as cans), as do the deposit rates. Generally, these systems lead to high return rates and a reduction of littering. On the other hand, the handling and administration costs can be substantial.

6.5 Case study

As we have seen in the preceding sections, deposit-refund schemes (DRS) can be effective instruments to redirect waste streams from final disposal to reuse and recycling. They reduce the incentive for illegal dumping (and thus the costs of monitoring and enforcement). On the other hand, the costs of handling and administration may be substantial.

Taking into account the 'priority waste streams' in Dutch waste policy, a number of products has been selected for an analysis of DRS impacts, focusing on waste streams for which a high degree of separate collection is considered to be important. This has in the following selection:

- Electric and electronic equipment (conversion of the current system of producer responsibility to a DRS scheme);
- Batteries (idem);
- Carpets, furniture and mattresses (taking into account current initiatives in the area of 'product-service combinations', in which the user 'leases' the product and returns it to the producer/supplier when it is discarded).

³⁸ For details by country, see Appendix I.

For batteries as well as electric and electronic equipment, the Netherlands introduced producer responsibility in 1995 and 1999, respectively. Producer responsibility implies that producers are responsible for the disposal of batteries and electric and electronic equipment. In the case of electric and electronic equipment, producers stimulate take back activities and separate collection of appliances. In the case of batteries, the separate collection is widely stimulated. In 2007, the amount of batteries separately collected was 38% of the amount sold, while 86% of the batteries disposed is separately collected (Stibat, 2008). For electric and electronic equipment, the share of appliances reused or recycled is 43% of the number of appliances sold in 2006. About two-third of all electric and electronic equipment disposed were separately collected (NVMP, 2007).

As mentioned in Chapter 2, a partial general equilibrium model developed by Fullerton and Wu (1998) is used for the estimation of impacts of introducing a DRS (see Appendix III for a detailed description of the model). In a DRS, products are taxed when consumers purchase the product, and the products are subsidized when they are separately disposed of. By imposing a tax on consumer products (i.e. batteries and small electric appliances) and a subsidy on recovery for recycling, we can evaluate a DRS system. For furniture, there is too little information to apply the Fullerton-Wu model, and therefore a qualitative analysis will be done.

6.5.1 Household batteries

Waste flows

As shown in Table 6.2, the total number of batteries sold grew with 72% in the period 2000-2008. The number of rechargeable batteries sold more than doubled in the same period. The share of rechargeable batteries increased from 9 to 12%.

Table 6.2 Development of the number of disposable and rechargeable batteries sold in the period 2000-2007 (in thousands per year).

	Disposable batteries	Rechargeable batteries	Total	Share of rechargeable batteries
2000	189.2	19.8	208.9	9.5
2001	204.4	18.8	223.2	8.4
2002	203.6	20.1	223.8	9.0
2003	244.5	27.9	272.4	10.3
2004	280.5	36.4	316.9	11.5
2005	309.2	38.9	348.1	11.2
2006	315.2	38.7	353.9	10.9
2007	317.6	42.4	360.0	11.8

Source: Stibat (2008)

As shown in Figure 6.1, the total amount of batteries sold increased with 34% in the period 2002-2007 and exceeded the amount of 8,000 tonnes by 2007. The number of appliances using batteries is still increasing (toys, cell phones, etcetera). The total amount of batteries separately collected increased with almost 50% and amounted to 3,015 tonnes in 2007, leaving still another 479 tonnes of batteries in household residual waste. The amount of batteries in household rest waste declined gradually between 2002 and 2007 with 40%.

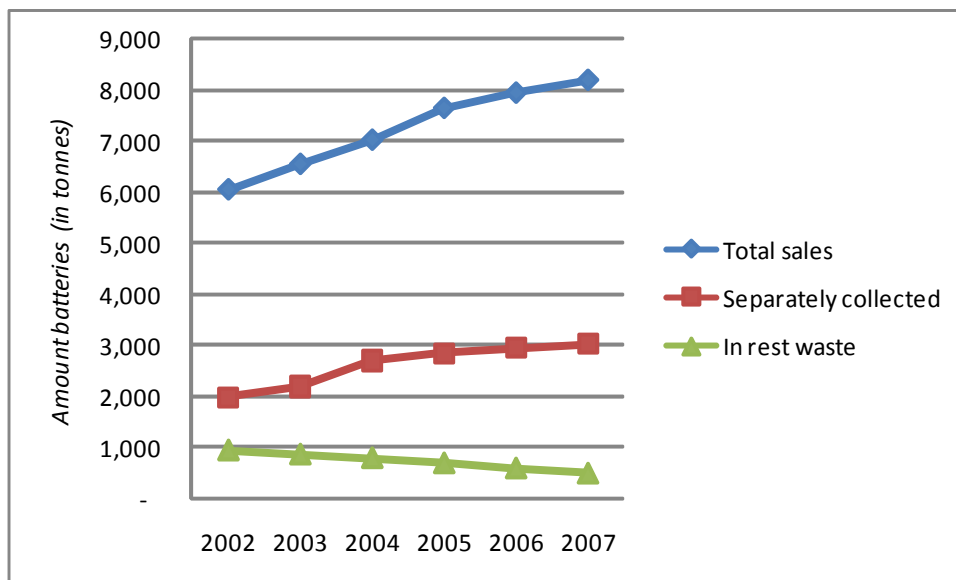


Figure 6.1 Development of batteries sold, collected and separated from rest waste, 2002-2007. Source: Stibat (2008)

Since 2004, 80% of the batteries collected were collected separately, which was the target (Stibat, 2008). In 2002, this was only 71%, while in 2007 it was 86%. However, from 2009 onwards, the recovery rate is no longer calculated based on the total amount of batteries collected but on the total amount of batteries sold according to the new EU Directive for batteries (2006/66/EC). The recovery targets from the EU Directive are 25% in 2012 and 45% in 2016.

In 2007, 3,015 tonnes of batteries in the Netherlands were collected separately, while the total weight of batteries sold was 8,215 tonnes, resulting in a recovery rate of 36.7%. There are a number of reasons for the discrepancy between sales and collection. First, new batteries are stored before usage. Second, batteries are stored after usage. Third, the lifetime of batteries increases due to the fact that the share of rechargeable batteries increased (i.e. 12% of the batteries sold are rechargeable).

*Results for Batteries*³⁹

With the adjusted Fullerton-Wu model, we simulate the impacts of the DRS system for batteries in the Netherlands. The base situation is that 3,015 tonnes of batteries have been collected separately, and 1,093 tonnes of batteries can still be recycled. From the Fullerton-Wu model, we derive the demand for the collection of rest waste and the demand for recovery. Both types of demand depend on the prices of rest waste collection and recovery including taxes, respectively, instead of the total amount of batteries sold. Imposing a deposit on the purchase of a commodity will reduce the demand for the commodity if the price elasticity is negative. The model further assumes that batteries can be reused or recovered for the production of new batteries. Also, firms pay a price for recovered batteries to households including the refund.

For the model, we assume that a maximum of half of the batteries sold will return after one year. The reasons to assume that the average lifespan of batteries is longer than one year are the same three as those just mentioned for the discrepancy between sales and collection.

In the literature, estimations of the price elasticity of the demand for batteries are not available. ECOTEC (2001) argues that the price elasticity of batteries is very low. Dimarso (1995) concluded in a conjoint analysis that consumers are hardly aware of the price of batteries. Therefore, we assume a -0.1-price elasticity of the price of new batteries on the generation of rest waste batteries and recovered batteries. If the price of batteries increases, the demand for batteries will decline, and consequently, the generation of rest waste batteries and recovered batteries as well. Following Dimarso (1995), we assume a -0.1-price elasticity of the price of recovered batteries on the generation of rest waste batteries. The own-price elasticity of recovered batteries is positive (0.15).⁴⁰ The higher the price of recovered batteries, the more households are willing to recover batteries. The actual values of the price elasticities are unknown, so we use conventional assumptions for the estimations. Since there is no convincing evidence for the low price elasticity, we also present results of DRS scenarios with a more elastic demand for batteries.

For a fixed rate of recyclability ($0.367=3.015/8.215$)⁴¹, we simulated the results for DRS systems with different levels of deposits and refunds. We simulated with deposits of €5, €10, €15 and €20 per kg. Given an average weight of 23.4 grams per battery (see Stibat, 2008), the '€5 per kg' deposit is equivalent to a deposit of €0.12 per battery, while a '€20 per kg' deposit is equivalent to €0.52 per battery. The average price per battery without deposits ranges from €1 to €2, so that the highest deposit considered is in

³⁹ For the analysis of the DRS system on batteries, we apply the Fullerton-Wu model (see the Appendix to this chapter). The model parameters are summarized in Table 6.10 in the Appendix to the present chapter.

⁴⁰ In practice, the price of recovered batteries is zero. In equilibrium models, however, we cannot assess prices that are initially zero. For convenience, we assume a positive price for recovered batteries. This price corresponds to the costs of manufacturers for using recycled products.

⁴¹ The recovery rate is calculated base upon recovered and sold batteries.

the range of 25-50% of the retail price of batteries. The labour input of the batteries production is constant.⁴²

Figure 6.2 shows the development of the recovery rate for batteries for deposits in the range of €5 to €20 per kg of batteries. Initially, the amount of batteries separately collected is 36.7% compared to the newly sold batteries. Due to a deposit of €5 per kg, the rate of separate collection would increase to 38.4% in the case that we assume that the price elasticities of recycling are low (inelastic). If the deposit rate is gradually increased to €20 per kg, the recovery rate would increase to 42.5%. This is equivalent to an 89.7% separate collection rate of all batteries disposed of. Note that it is assumed that the current widespread network of batteries collection is maintained.

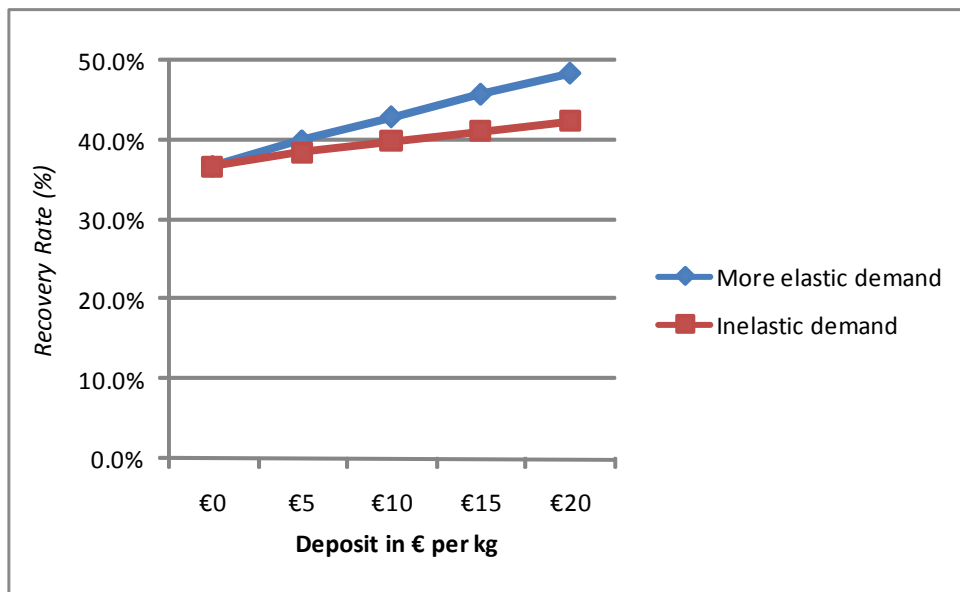


Figure 6.2 Changes in recovery rate of batteries when demand for batteries is inelastic and more elastic

Table 6.3 presents the results of the Baseline, DRS5, DRS10, DRS15 and DRS20 scenarios. The total amount of batteries in rest waste decreases with the implementation of a DRS system, while the amount of recovered batteries increases. In the case of a deposit of €20 per kilogramme (i.e. €0.511 per battery), the total amount of recovered batteries increases from 86.9 to 89.7% of the amount of batteries disposed.

⁴² In order to ‘close’ the partial equilibrium model, we assume that the labour input of the production of batteries is kept constant. Due to higher amounts of recovery, manufacturers of batteries will be able to produce more batteries.

Table 6.3 Results on DRS systems for batteries with an *inelastic demand**

Parameter values	Base	DRS	DRS	DRS	DRS
Deposit tax and refund (€ per appliance)	€0	€5	€10	€15	€20
<i>Changes in physical flows</i>					
- Batteries sold (tonnes)	8,219	8,258	8,293	8,323	8,349
- Batteries in rest waste (tonnes)	456	440	427	415	406
- Recovered (tonnes)	3,019	3,175	3,313	3,435	3,545
- Total amount of batteries waste (tonnes)	3,475	3,615	3,740	3,850	3,951
<i>Recovery rate</i>					
- Recovered batteries/Sold batteries (in %)	36.7	38.4	39.9	41.3	42.5
- Recovered batteries/Disposed batteries (in %)	86.9	87.8	88.6	89.2	89.7
<i>Changes in retail values</i>					
- Purchase price of batteries (€ per tonne)	48.7	48.4	48.2	48.1	47.9
- Total price of batteries (€ per tonne)	48.7	53.4	58.2	63.1	67.9
<i>Changes in recovery values</i>					
- Price of recovered batteries paid by firms (€ per tonne)	13.3	12.6	12.1	11.6	11.3
- Total price of recovered batteries received by households (€ per tonnes)	13.3	17.6	22.1	26.6	31.3

* In the model, the price of recovered batteries is equivalent to the costs of recovered batteries for the production of new batteries. Manufacturers pay the suppliers of recovered batteries (households). By supplying more recovered batteries, households can increase their budget and consequently purchase more new batteries. Simultaneously, the production and consumption of new batteries increases with the growing amount of recovered batteries given a fixed amount of labour input.

The total amount of batteries collected increases with 4.0% in the DRS5 scenario, while the total amount of batteries recovered increases with 5.2%. The prices for batteries including the tax of €5 per kg increased with 9.8%, while the price for recycling including the subsidy increased with 32.8% in the DRS5 scenario. The total amount of batteries sold increases with 0.5% due to increased revenues of recovered batteries for households (who spend their additional budget on new batteries). The retail prices excluding taxes declined with 0.5% to partially compensate the DRS imposed. Households face the price including tax, and firms receive the price without tax. Firms thus compensate part of the tax in order to reduce the decline in batteries sold.

According to a number of studies, the price elasticity of new batteries is low. However, there is no quantitative estimation of this price elasticity. In the inelastic demand scenario, we assumed that the price elasticity was -0.1. To verify the impact of the price elasticity, we now rerun the series of scenarios with a doubled price elasticity of the demand for new batteries, and for recovered batteries, see Table 6.10. The impact of a more elastic demand on the recovery rate is shown in Figure 6.2. Clearly, the amount of recovered batteries increases with a more elastic demand for batteries. In the case of a deposit of €20 per kg, the recovered rate would increase to 48.4% compared to 42.5% in the inelastic scenario. This means that 91.8% of the disposed batteries are separately collected (see Table 6.4).

Table 6.4 Results on DRS systems for batteries with a more elastic demand

Parameter values	Base	DRS	DRS	DRS	DRS
Deposit tax and refund (€ per appliance)	€0	€5	€10	€15	€20
<i>Changes in physical flows</i>					
- Batteries sold (tonnes)	8,215	8,292	8,359	8,418	8,471
- Batteries in rest waste (tonnes)	456	427	402	382	365
- Recovered (tonnes)	3,019	3,311	3,589	3,851	4,098
- Total amount of batteries waste (tonnes)	3,475	3,738	3,991	4,233	4,463
<i>Recovery rate</i>					
- Recovered batteries/Sold batteries (in %)	36.7	39.9	42.9	45.7	48.4
- Recovered batteries/Disposed batteries (in %)	86.9	88.6	89.9	91.0	91.8
<i>Changes in retail values</i>					
- Purchase price of batteries (€ per tonne)	48.7	48.2	47.9	47.5	47.2
- Total price of batteries (€ per tonne)	48.7	53.2	57.9	62.5	72.2
<i>Changes in recovery values</i>					
- Price of recovered batteries paid by firms (€ per tonne)	13.3	12.1	11.1	10.4	9.8
- Total price of recovered batteries received by households (€ per tonnes)	13.3	17.1	21.1	25.4	34.8

In conclusion, the impacts of the implementation of a DRS system for batteries are modest even if the deposit is set at €20 per kg. The share of separately collected batteries compared to the newly sold batteries increases from 36.7% to 42.5% (or to 48.4% if the demand for batteries is more elastic). The reason for this modest increase is probably the high share of batteries separately collected compared to the amount of batteries disposed. Almost 90% of the batteries disposed are separately collected with a more elastic demand.

The discrepancy between the amount of batteries sold in 2007 (8.2 million kg) and the amount of batteries disposed (3.5 million kg) is quite large. As shown in Figure 6.1, this discrepancy seems to be of a structural nature. On the one hand, the use of batteries increased due to the number of electr(on)ic appliances using batteries. On the other hand, the storage of unused and/or used batteries in households increased. Unfortunately, there is no information on storage of used or unused batteries at households.

The implementation of a DRS system for batteries would also induce higher transaction costs, which were not accounted for in the model. Especially in the case of batteries, where there is already a wide network of free batteries collection, the transaction costs of implementation could be limited if the current network would be maintained. However, because collectors have to return the refund for batteries, a DRS system will inevitably lead to higher transaction costs for collectors. This might lead to a partial breakdown of the collection network, unless the recycling of batteries is economically attractive for both recyclers and collectors.

6.5.2 Small household appliances

Waste flows

In 2007, there has been an extensive survey on the possession, purchase and disposal of electric household appliances (GFK Panel Services Benelux, 2007). This section concentrates on white and brown goods, and specifically on the small white and brown goods that often end up in the households' rest waste. White goods are appliances used in the kitchen and used for personal care. Examples of large white goods are refrigerators, freezers, washing machines and dishwashers. Small white goods are all kinds of kitchen appliances, such as coffee machines and blenders, and appliances for personal care, such as shaving equipment, electric toothbrushes. Brown goods are for example TV (visual), radio and CD player (home video) and portable appliances.

Table 6.5 presents the disposal of the different white and brown good appliances. The disposed appliances are either reused directly (through second-hand shops or virtual second hand markets on the Internet like E-bay), or recycled. For recycling, we assume that the disposal of appliances through separate collection or large household waste collection makes recycling feasible. The discarded appliances are disposed in the rest waste. More than half of all brown and white goods are either reused or recycled. For the small white goods, 70% of the collected appliances is recycled or reused. More than 20% of the small white goods is disposed of as rest waste.

The number of discarded brown goods is smaller (4.4 million) than the number of discarded white goods (13.8 million) (see Table 6.6). The recovery rates of 57% for portable brown goods and other brown goods are the lowest for brown goods categories. In the case of the portable brown goods, 30% of the appliances are disposed of as households' rest waste. For small appliances, such as small white goods and portable brown goods, it is relatively easy to dispose of appliances as household rest waste. A DRS system can be effective in reducing these disposed appliances in households' rest waste. The main reason for discarding white and brown goods is a deficiency of the appliance (49% for brown goods and 56% for white goods), under-utilisation (21% and 17%) and the purchase of a newer, better quality version of the appliance (16 and 14%).

Table 6.5 Possession, purchase and disposal of white and brown good appliances (in millions pieces in 2006)

	Possession	Purchased – new	Purchased – Second hand	Disposed	Share of reuse
Large white goods	45.4	3.7	0.9	3.1	34
Small white goods	167.0	19.6	1.3	10.7	27
- Kitchen	83.7	9.7	0.8	5.5	31
- Personal care	37.9	5.2	0.1	2.5	19
- Others	45.5	4.7	0.4	2.7	27
Brown goods	102.9	10.1	1.4	4.4	36
- Visual	38.5	3.5	0.6	1.4	35
- Home audio	22.0	1.0	0.3	0.8	46
- Portable	30.4	3.6	0.2	1.6	31
- Other brown goods	12.0	2.0	0.3	0.6	41

Source: GFK Panel Services Benelux (2007)

Table 6.6 *Disposal, reuse and recycling of white and brown good appliances (in million pieces in 2006)*

	Total waste	Reused	Recycled*	Disposed of as rest waste	Other	Recovery rate**
Large white goods	3.1	1.0	1.6	0.3	0.2	85
Small white goods	10.7	2.9	3.6	3.2	1.0	61
- Kitchen	5.5	1.7	1.8	1.5	0.5	63
- Personal care	2.5	0.5	0.8	1.0	0.2	51
- Others	2.7	0.7	1.0	0.7	0.3	64
Brown goods	4.4	1.6	1.3	0.8	0.6	67
- Visual	1.4	0.5	0.6	0.2	0.1	77
- Home audio	0.8	0.4	0.2	0.1	0.1	75
- Portable	1.6	0.5	0.4	0.5	0.2	57
- Other brown goods	0.6	0.2	0.1	0.1	0.2	61

Source: GFK Panel Services Benelux (2007) recalculated by authors.

* The recycled appliances are separately collected, see GFK Panel Services Benelux (2007)

** Based on the calculation (The sum of the numbers of reused and recycled appliances divided by total number of disposed appliances)

Results for small white goods⁴³

For the scenario for brown and white goods, we concentrate on a DRS system for small white goods such as small kitchen appliances, small appliances for personal care and other small ‘white good’ appliances. In the Netherlands, households possess 167 million small white goods (i.e. 10 appliances per household). In 2006, 10.7 million small appliances were discarded, of which 2.9 million appliances were reused. The main reason why households (i.e. 57% of the cases) dispose small white appliances is that the appliance broke down (GFK Panel Services Benelux, 2007). Another 3.6 million appliances were separately collected through take-back activities of stores or waste collection centres. The rest of the small appliances end up in the rest waste.

The life span of small white appliances is more than 10 years (GFK Panel Services Benelux, 2007). With the adjusted Fullerton-Wu model, we simulate the impacts of the DRS system for small white appliances. The base situation is that 3,6 million appliances are recovered for recycling (i.e. collected separately), 2,9 million appliances reused and 4,2 million appliances could still be recycled (including so called others in Table 6.6).

We use the same assumptions for the Fullerton-Wu model as the previous subsection on batteries. Both the demand for the collection of rest waste and the demand for recovery depend on the prices of rest waste collection, the prices of recovery, and the prices of new small white appliances including taxes and deposits. Imposing a deposit on the purchase of an appliance will reduce the demand for the appliance if the price elasticity is negative. The model further assumes that recovered appliances can be reused or recycled for the production of new appliances. Also, firms pay a price for recovered appliances to households including a refund.

⁴³ For the analysis of the DRS system on appliances, we apply the Fullerton-Wu model. The model parameters are summarized in Table 6.11 in the Appendix to this chapter.

Similar to batteries, there is no specific information on the price elasticity of demand for small white appliances available in the literature. Since the lifespan of appliances is more than 10 years, the income elasticity will play a larger role in the demand for appliances than price elasticities. Therefore, we assume a similar price elasticity for small appliances as was used for batteries: -0.1-price elasticity of the price of new appliances on the generation of rest waste appliances and recovered appliances. If the price of a new appliance increases, the demand for new appliances will decline, and as a consequence, the generation of rest waste and recycling declines as well. In addition, we assume that a -0.1-price elasticity of recovered appliances on the generation of rest waste appliances. The own-price elasticity of recovered appliances is positive (0.15).⁴⁴ The higher the price of recovered appliances, the more household are willing to recover appliances. The actual values of the price elasticities are unknown, so we use conventional estimates for the estimations, i.e. the 'inelastic demand scenario' as shown was in Figure 6.1. Since there is no evidence for price elasticities for the demand of appliances, we also present results of DRS scenarios with a more elastic demand for appliances.

We simulate the impact of the DRS system with different levels of deposit rates. A similar subsidy is provided for recovered appliances. The average price per appliance varies with the type of appliance. For convenience, we set an average price at €25, and simulate the impact of three tax scenarios: €5, €10 and €15 per appliance.

For the production of small white appliances, the labour input supplied by households is kept constant. In order to 'close' the partial equilibrium model, we assume that the labour input is fixed. As a consequence, an increase in recovered appliances by households will stimulate the production of small white appliances.

Figure 6.3 shows the development of the recycling rate for small white appliances for taxes in the range of €5 to €15 per appliance. Initially, the recovery rate (amount of small white appliances separately collected and reused or recycled as a percentage of the total amount sold) is already 41.6%. Due to a deposit rate of €5, the rate of separate collection would increase to 45.5% in the case we assume that the price elasticities of recovered appliances are low (inelastic). If the deposit rate is gradually increased to €15, the recovery rate would increase to 50.8%, which is equivalent to 70% of all small white appliances disposed would be separately collected, see Table 6.7. The model assumes that the increased recovery of appliances implies that more recovered products can be used in the production of small white appliances. The prices of appliances without deposit would slightly decrease.

⁴⁴ In practice, the price of recovered small white appliances is zero. In equilibrium models, however, we cannot assess prices that are initially zero. For convenience, we assume a positive price for recovered small white appliances. This price corresponds to the costs of manufacturers for using recovered products.

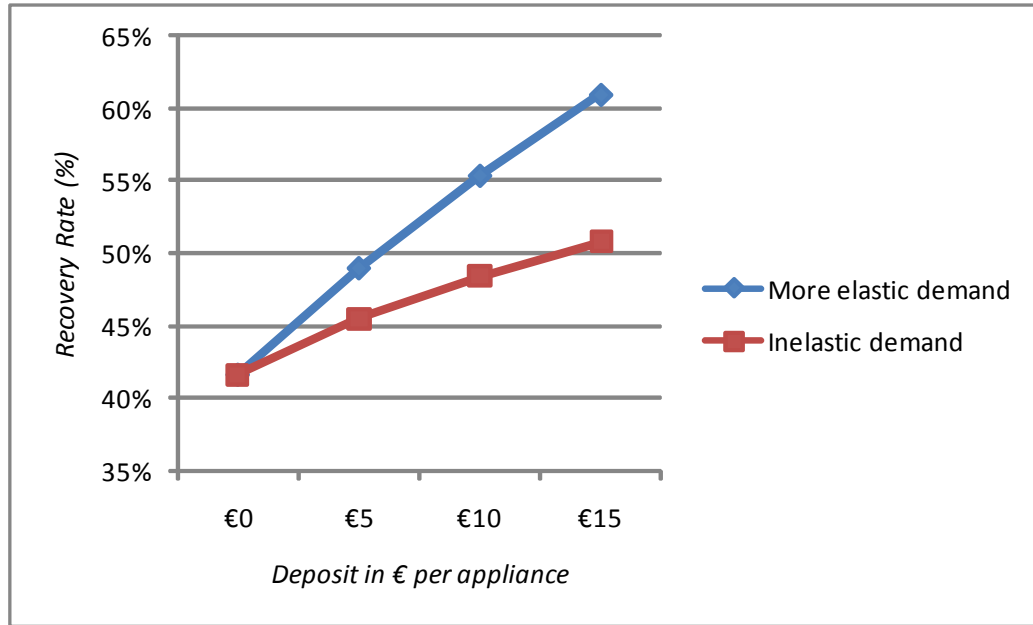


Figure 6.3 Changes in recovery rate of small white appliances when demand for small white appliances is inelastic and more elastic

Table 6.7 presents the results of the Baseline, DRS5, DRS10 and DRS15 scenarios. The total amount of appliances in rest waste decreases with the implementation of a DRS system, while the amount of recovered appliances increases. In the case of a deposit rate of €15 per appliance, the total amount of recovered appliances is more than 60% of the amount of new appliances sold. In addition, almost 70% of the disposed appliances are then recovered.

As mentioned above, we have no information from the literature on the price elasticity of small white appliances. For the results of the original series of scenarios in Table 6.4, we assumed an inelastic demand for small white appliances. However, we rerun the series of simulations with double price elasticities for the demand for small white appliances and for recovered appliances (see

Table 6.11). The impact on the recovery rate of a more elastic demand for small white appliances for the various DRS levels is shown in Figure 6.3 and Table 6.8. The recovery rates increase substantially more if the price elasticity of demand is doubled. Under these conditions, the total amount of appliances in rest waste decreases with the implementation of a DRS system, while the amount of recovered appliances increases more. In the case of a tax of €15 per appliance, the total amount of recovered appliances is more than 60% of the amount of new appliances sold. In addition, more than three quarter of the disposed appliances is then recovered.

Table 6.7 Results on DRS systems for small white appliances with an inelastic demand

Parameter values	Base	DRS	DRS	DRS
Deposit tax and refund (€ per appliance)	€0	€5	€10	€15
<i>Changes in physical flows</i>				
- Appliances sold (millions)	15.60	15.74	15.86	15.95
- Appliances in rest waste collected (millions)	4.21	3.92	3.72	3.58
- Recovered (millions)	6.49	7.16	7.68	8.10
- Total amount of appliances disposed (millions)	10.70	11.08	11.40	11.68
<i>Recovery rate</i>				
- Recovered appliances/Sold appliances (in %)	41.6	45.5	48.4	50.8
- Recovered appliances/Disposed appliances (in %)	60.7	64.7	67.3	69.4
<i>Changes in retail values</i>				
- Purchase price of appliances (€ per appliance)	25.64	25.39	25.22	25.08
- Total price of appliances (€ per appliance)	25.64	30.39	35.22	40.08
<i>Changes in recycling values</i>				
- Price of recovered appliances paid by firms (€ per appliance)*	6.16	5.58	5.21	4.94
- Total price of recovered appliances received by households (€ per appliance)	6.16	10.58	15.21	19.94

* In the model, the price of recovered appliances is equivalent to the costs of recovered appliances for the production of new appliances. Manufacturers pay the suppliers of recovered appliances (households). By supplying more recovered appliances, households can increase their budget and consequently purchase more new appliances. Simultaneously, the production of new appliances increases with the increased amount of recovered appliances given a fixed amount of labour input.

Table 6.8 Results on DRS systems for small white appliances with a more elastic demand

Parameter values	Base	DRS	DRS	DRS
Deposit tax and refund (€ per appliance)	€0	€5	€10	€15
<i>Changes in physical flows</i>				
- Appliances sold (millions)	15.60	15.89	16.10	16.28
- Appliances in rest waste collected (millions)	4.21	3.69	3.34	3.08
- Recovered (millions)	6.49	7.79	8.92	9.94
- Total amount of appliances disposed (millions)	10.70	11.48	12.26	13.01
<i>Recovery rate</i>				
- Recovered appliances/Sold appliances (in %)	41.6	49.0	55.4	61.0
- Recovered appliances/Disposed appliances (in %)	60.7	67.9	72.8	76.4
<i>Changes in retail values</i>				
- Purchase price of appliances (€ per appliance)	25.64	25.18	24.84	24.57
- Total price of appliances (€ per appliance)	25.64	30.18	34.84	39.57
<i>Changes in recycling values</i>				
- Price of recovered appliances paid by firms (€ per appliance)*	6.16	5.13	4.48	4.03
- Total price of recovered appliances received by households (€ per appliance)	6.16	10.13	14.48	19.03

The scenarios with the adjusted Fullerton-Wu model show that the implementation of a DRS system has a significant positive impact on the recycling rates. In the case of small white appliances, the income effect of the demand for small white appliances is ignored, although income effects play a significant role in the purchase of durable goods. The underlying assumption of the Fullerton-Wu model for small white appliances is that the present system of collection due to the producer responsibility will be maintained. Especially in the case of appliances, where there is already a wide network of free appliances collection (as part of the producer responsibility), the transaction cost of a collection system is probably limited. However, a DRS system might lead to higher transaction costs for collectors, because they have to return the refund for all individual appliances.

Finally, we aim at translating the above results into an environmental assessment. However, the literature does not provide us with a clear workable indicator of the environmental impact due to the fact that electric appliances include many different types of materials, and the white and brown goods cover a wide range of different appliances that are produced by different manufacturers. Therefore, we conclude that, in general, the increment of reuse and recycling of electric appliances is considered to be positive for the environment.

6.5.3 Carpets, mattresses, furniture

Waste flows

On the carpets, mattresses and furniture, there is no centralised information on the numbers or amounts sold. In fact, the only source we do have is the amount of carpets, mattresses and furniture disposed of by households. This information will not give a clear picture on how many carpets, mattresses and pieces of furniture are disposed of,

because there is no information on the numbers that are disposed of through second hand shops, or through the virtual second hand market on the Internet.

As shown in Table 6.9, 13 Ktonnes of carpets and 35 Ktonnes of furniture were collected in 2005. In the case of carpets, most of them were brought to a disposal point, while 75% of the furniture was collected by waste collectors. Note that both carpets and furniture were separately collected. How many carpets and furniture are resold via second-hand shops or via the Internet is unknown. In addition, there is no readily available information on the sales of carpets, furniture and mattresses.

Table 6.9 Carpets and Furniture collected from households, 2005(in Ktonnes)

	Collected at the curb	Brought to community waste center	Total
Carpets	1	12	13
Furniture	26	9	35

Source: CBS/SenterNovem (2008)

Results carpets, mattresses and furniture

Since there is no or little information on carpets, furniture and mattresses, we cannot apply the Fullerton-Wu model as was done for small white appliances or batteries. However, the case of carpets, furniture and mattresses to a fair degree resembles the case of small white appliances because of comparable characteristics of the waste flows. Similar to small white appliances, carpets, furniture and mattresses have an active second-hand market and generally have a long lifespan (ten years or more). In terms of environmental impact, the existence of a second-hand market is positive, because the lifetime of carpets, furniture and mattresses can be extended in this way. The difference between small white goods and carpets, furniture and mattresses waste streams is that small white appliances can be disposed of in rest waste (containers or bags), while carpets, furniture and mattresses cannot due to their size.

The impact of the implementation of a DRS system for carpets, furniture and mattresses is ambiguous. On the one hand, a DRS system could support the separate collection of these waste types while simultaneously discouraging their storage. After all, the refund encourages households to bring back unused items. In this way, the recycling of these products is stimulated as well.

On the other hand, the provision of a refund can have a negative effect on the lifetime of carpets, furniture and mattresses. A refund can discourage households to sell carpets, furniture and mattresses on a second-hand market. In particular, if the market price on the second-hand market is lower than the refund, households will dispose of the good. If the market price is only slightly higher than the refund, the choice between reselling or disposing depends on the effort households have to make to resell the good. In addition, since the lifespan of carpets, furniture and mattresses is long, the implementation of a DRS system will take a long period before it is effectively implemented.

Note that the characteristics of carpets, furniture and mattresses are different. In particular, furniture is often suitable for reselling, which is often much more difficult with carpets and mattresses.

6.6 Conclusions

Deposit-refund scheme (DRS) can be effective in redirecting waste streams from final disposal to reuse and recycling. The advantage of DRS is that it reduces the incentive of illegal dumping while it simultaneously stimulates reuse and recycling of products. In addition, it reduces the amount of waste. For large bottles, with a recovery and recycling rate of 95%, the voluntary DRS in the Netherlands is very successful. Foreign experiences with mandatory DRS show that these systems also tend to lead to high return rates and a reduction of littering. On the other hand, the handling and administration costs can be substantial.

The success of a DRS depends largely on the existing design of the system from which it has to emerge. For example, as a result of the principle of producer responsibility, a comprehensive collection system is present in the Netherlands for batteries and small white appliances. The added value of a DRS system for batteries in terms of increased separate collection will be modest, as the present rate of separate collection is already high. For small white appliances more added value can be expected from the introduction of a DRS system.

Small white appliances as well as carpets, furniture, and mattresses are characterized by lifetimes of ten years or more. This implies that the time period between deposit and refund is relatively long. The introduction of a full DRS will take ten years or more. Moreover, for small white appliances as well as carpets, furniture, and mattresses, there is an active (physical and virtual) second-hand market with which the lifetime of the products can be extended. A DRS will probably have a negative effect on these second-hand markets, and thus on the lifetimes of the products. The environmental impacts of the effects of a DRS remain ambiguous and therefore require further research.

Appendix of Chapter 6

Parameter values for batteries and small white appliances

Table 6.10 Initial values for variables and parameter values for the batteries example of the adjusted Fullerton-Wu model

Variable/ parameter	Definition	Low price elasticities (Base, DRS5 and DRS20 scenarios)	Moderate price elasticities (DRS5' and DRS20' scenarios)
<i>Overall model assumptions</i>			
q	Composite commodity	8,125	8,125
ρ	Degree of recoverability (=3.015/8.215)	0.367	0.367
θ	Packaging rate	0.05	0.05
r	Recovery	3,015	3,015
g	Garbage collection	1,093	1,093
k_q	Inputs of resources for production of f	167.5	167.5
k_g	Inputs of resources for production of g	10	10
h or k_h	Commodity produced and consumed at home	10	10
<i>Waste and recovery sector prices</i>			
p_r	Price of recovered goods	13.3	13.3
p_q	Price of commodity goods	48.7	48.7
p_g	Price of garbage	9	9
<i>Rest waste generation (households)</i>			
δ_0	Scaling parameter	1.30	2.48
δ_ρ	Parameter for recoverability	-0.2	-0.2
δ_θ	Parameter for packaging	0.2	0.2
δ_{pq}	Parameter for commodity price	-0.1	-0.2
δ_{pr}	Parameter for recovery price	-0.1	-0.2
<i>Recovery production (households)</i>			
γ_0	Scaling parameter	1.70	0.78
γ_ρ	Parameter for packaging	0.2	0.2
γ_{pq}	Parameter for commodity price	0.1	0.2
γ_{pr}	Parameter for recovery price	0.15	0.3
<i>Production function (firms)</i>			
β_0	Scaling parameter	2.01	2.01
β_k	Parameter for resource inputs	0.5	0.5
β_r	Parameter for recovered inputs	0.1	0.1
β_ρ	Parameter for recoverability	-0.1	-0.1
β_θ	Parameter for packaging	0.5	0.5
E	Garbage production parameter	0.0457	0.0457

Table 6.11 Initial values for variables and parameter values for the appliances example of the adjusted Fullerton-Wu model

Variable/ parameter	Definition	Low price elasticities (Base, DRS5 and DRS20 scenarios)	Moderate price elasticities (DRS5' and DRS20' scenarios)
<i>Overall model assumptions</i>			
q	Composite commodity	15.6	15.6
ρ	Degree of recoverability (=6.49/15.6)	0.416	0.416
θ	Packaging rate	0.1	0.1
r	Recovery	6.5	6.5
g	Garbage collection	4.2	4.2
k_q	Inputs of resources for production of f	140	140
k_g	Inputs of resources for production of g	10	10
h or k_h	Commodity produced and consumed at home	20	20
<i>Waste and recovery sector prices</i>			
p_r	Price of recovered goods	6.2	6.2
p_q	Price of commodity goods	25.6	25.6
p_g	Price of garbage	2.4	2.4
<i>Rest waste generation (households)</i>			
δ_0	Scaling parameter	9.3	15.4
δ_ρ	Parameter for recoverability	-0.2	-0.2
δ_θ	Parameter for packaging	0.2	0.2
δ_{pq}	Parameter for commodity price	-0.1	-0.2
δ_{pr}	Parameter for recovery price	-0.1	-0.2
<i>Recovery production (households)</i>			
γ_0	Scaling parameter	4.3	2.3
γ_ρ	Parameter for packaging	0.2	0.2
γ_{pq}	Parameter for commodity price	0.1	0.2
γ_{pr}	Parameter for recovery price	0.15	0.3
<i>Production function (firms)</i>			
β_0	Scaling parameter	8.7	8.7
β_k	Parameter for resource inputs	0.25	0.25
β_r	Parameter for recovered inputs	0.2	0.2
β_ρ	Parameter for recoverability	-0.05	-0.05
β_θ	Parameter for packaging	0.6	0.6
E	Garbage production parameter	0.421	0.421

The Fullerton-Wu model

The Fullerton-Wu model distinguishes a representative household that consumes commodities q and either recovers products r or disposes products after usage g . The production of garbage depends on the total amount of consumption commodities q , and the packaging rate θ and the degree of recoverability of the commodities ρ . In addition to the Fullerton-Wu model, we assume that the generation of disposed product also depends on the price level of the consumption commodity p_q and the price level of recovered products p_r .

The technology of waste disposal is:

$$g = \delta_0 \rho^{\delta_\rho} \theta^{\delta_\theta} (p_q + t_q)^{\delta_{pq}} (p_r + t_r)^{\delta_{pr}}, \quad \text{III.1}$$

where the δ 's are the parameters of the variables and δ_0 is the positive scaling parameter. The parameter δ_θ is positive, which means that more commodities and a higher share of packaging of the commodities will lead to more generation of waste disposal. We assume that the total amount of consumption commodities remains constant. Waste disposal is restricted to the products considered. Higher price levels for commodities or recovery (including taxes t) will lead to lower levels of waste disposal, because the demand for new commodities will decline (and waste disposal as well) and recovery becomes more attractive. The technology of recovery by households is:

$$r = \gamma_0 \rho^{\gamma_\rho} (p_q + t_q)^{\gamma_{pq}} (p_r + t_r)^{\gamma_{pr}}, \quad \text{III.2}$$

where the γ 's are the parameters of the variables and γ_0 is the positive scaling parameter. The parameter for ρ is positive, while the parameters for the commodity price (γ_{pq}) and recovery price (γ_{pr}) are negative. Higher price levels will lead to more recovery.

Utility of households depends on the amount of consumer commodities q , and the amount of recovery and other commodities GN . Households maximize utility subjected to a budget constraint. The utility function is:

$$U = q^{\alpha_q} h^{\alpha_h} r^{\alpha_r} GN^{\alpha_{GN}}, \quad \text{III.3}$$

where α 's are the utility parameters which are all positive. The variable h is the amount of resources produced and consumed at home, such as time spent on recovery for instance. More recovery efforts at home (higher level of h) will result in higher level of utility. The budget constraint is:

$$(k - k_h) + (p_r + t_r)r = (p_q + t_q)q + (p_g + t_g)g, \quad \text{III.4}$$

where k is the resource inputs of firms to produce commodities which are supplied by households, k_h is the resource inputs used at home ($k_h = h$), t_r is the subsidy on recovery, t_q is the tax on commodities, p_g is the price of waste collection and t_g is the tax of waste collection. The left-hand side of Eq. III.4 reflects the revenues from labour inputs and recovery, while the right-hand side reflects the expenditures on consumption and waste collection.

Firms produce commodities and have the option to use recovered products or the packaging rate of the commodities. The production function is:

$$q = \beta_0 k_q^{\beta_k} r^{\beta_r} \rho^{\beta_\rho} \theta^{\beta_\theta}, \quad \text{III.5}$$

where k_q is the resource used for production (labour), and β 's are the production function parameter. The parameter β_0 is the scaling parameter. The production function exhibits constant returns to scale, so that $\beta_k + \beta_r + \beta_\rho + \beta_\theta = 1$. Firms maximize profits (π), and the profit function is:

$$\pi = p_q q - p_r r - k_q. \quad \text{III.6}$$

In addition, there is a waste collection industry with the linear production technology:

$$g = \varepsilon k_g \quad \text{III.7}$$

where k_g is the resource input for the waste collection industry provided by households.

The parameter ε reflects the amount of resource input per unit of waste collected. To close the model, the sum of resource inputs is limited to k , so that $k = k_q + k_h + k_g$.

7. Subsidies and fiscal incentives

7.1 Introduction

Subsidies can be used to give a positive financial incentive to producers and consumers who perform certain behaviour that is considered to be socially beneficial from the government's point of view. As such, a subsidy is the 'mirror image' of a tax or charge: it internalises positive rather than negative externalities. In this chapter, we will analyse some illustrative examples of using subsidies in waste policy. We use a wide interpretation of the concept of a subsidy, including preferential public procurement (under which public authorities are prepared to pay a higher price for environmentally beneficial products and services) as well as environmentally motivated tax reductions.

7.2 Theory and literature

Although subsidies in general conflict with the 'polluter pays principle', they may still be useful policy instruments in situations where market imperfections (such as transaction and monitoring costs) preclude the use of 'first best' tools such as waste disposal fees. For example, Dinan (1993) and Eichner (2005) conclude that the optimum instrument package should include subsidies on reuse or recycling. Likewise, Fullerton and Wu (1998) show that, if market signals cannot be corrected by means of disposal charges, welfare can be improved by subsidies to recycling or recyclability (in combination with product taxes).

Subsidies can have high administrative costs if newly introduced as a 'stand alone' instrument. Significant cost reductions can be achieved if the subsidy is linked to (fiscal) schemes that already exist, e.g. the Value Added Tax (VAT) or income and profit taxes.

7.3 The Dutch situation

In addition to the taxes and charges that have been discussed in the previous chapters, Dutch waste policy also applies positive financial incentives such as subsidies and fiscal facilities. For domestic, commercial and office waste the following schemes are particularly relevant.

Waste Fund

In July 2007 a framework agreement was signed between the Ministry of Environment (VROM), the Association of Dutch Municipalities (VNG) and the packaging industry, on the establishment of a Waste Fund ('Afvalfonds'). Annually an amount of EUR 115 million from the VROM budget will be paid to the Waste Fund. All municipalities receive an amount from the Waste Fund for the separate collection of packaging waste from households. This amount consists of a guaranteed price and a remuneration for transport costs. In fact this arrangement is a 'voluntary fiscalisation' of the producer responsibility for packaging. For practical reasons it was decided to leave the execution to the municipalities and organise the financing through a surcharge on the packaging tax. The costs of the 'Impulse Programme on Littering' (EUR 11 million per year) are also financed from the Waste Fund.

Subsidy scheme for dealing with litter (SAZ)

The 'Subsidy scheme for dealing with litter' ('Subsidieregeling aanpak zwerfafval', SAZ) is part of the overarching 'Impulse Programme on Littering'. Its objective is to make the Netherlands 'visibly and measurably cleaner' within a period of three years (2007 through 2009). The SAZ contains three types of projects. 'Basic' and 'plus' projects can be only applied for and carried out by municipalities; 'test field' projects are open for everybody who wants to establish or test innovative ideas concerning dealing with litter. The total budget of the SAZ amounts to EUR 15.9 million.

Subsidy scheme for sustainable energy production (SDE)

The subsidy scheme for sustainable energy production ('Stimuleren Duurzame Energieproductie', SDE) compensates the difference in production costs between fossil and sustainable energy. Among the types of sustainable energy for which SDE subsidy is available are electricity from waste incineration and biogas from the digestion of organic waste. For waste incineration plants the subsidy amount is dependent on the energetic efficiency, which should be at least 22%. For energy from organic waste the basic subsidy amount is EUR 0.12 per kWh. The actual electricity price (to be determined annually) is subtracted from the basic amount.

Fiscal facilities for firms (MIA and Vamil)

The 'MIA' and 'Vamil' schemes offer a fiscal advantage to enterprises investing in environmentally friendly equipment. 'MIA' allows for an additional deduction opportunity from fiscal profit. When the firm has purchased a piece of equipment that is on the 'Environmental List', he can deduct (in addition to the standard depreciation) 15, 30 or 40% of the investment costs from the profit in the year in which the investment was made. Under the 'Vamil' scheme the firm can freely determine when he wants to depreciate the investment costs of the means of production. Early depreciation can be advantageous in terms of liquidity and interest, because tax payments are postponed.

The 'Environmental List' is subject to annual review. For some means of production 'MIA' and 'Vamil' can be combined; for others only one of these is applicable. The 'Environmental List' for 2008 contains several kinds of equipment that are aimed at resource conservation, waste prevention, reuse, recycling or environmentally friendly waste management.

Programme Environment and Technology

The Programme Environment and Technology (Programma Milieu & Technologie) stimulates Dutch industry to develop and apply innovative processes, products and services. Its target group are industrial small and medium-sized enterprises (SMEs). Support from this programme is currently given, among others, to the development of a compression system for domestic waste.

7.4 Foreign experiences⁴⁵

Financial support from the government budget to activities that contribute to waste reduction and better waste management is given in a variety of ways across countries. Examples include direct grants (for R&D or investments), conditional money transfers to lower authorities, ‘soft’ loans, and tax reductions.

7.5 Case study Subsidies

Under certain conditions, the use of subsidies can be a justifiable exemption to the general ‘polluter pays principle’ in environmental policy. Among these conditions is the case in which it is impossible to internalise the full external costs of a product over its entire lifecycle. Food is a case in point. Food prices are kept artificially low, among others by means of reduced VAT rates. Moreover, farmers do not pay for most of the environmental damage caused by their activities (such as the eutrophication of surface and ground waters, and the contribution of livestock’s CH₄ and N₂O emissions to global warming). As a result (and despite the recent price increases on the world market) food remains ‘too cheap’, providing incentives for wasteful behaviour. In the Netherlands, consumers throw away food with an estimated value of between EUR 600 million and 1.6 billion per year, and another EUR 900 million to 2 billion is wasted in the ‘food chain’ (Meeusen and Hagelaar, 2008).

Given the fact that full application of the ‘polluter pays principle’ in agriculture is politically or practically not feasible, a ‘second best’ option to stimulate the reduction of food wastage could be to subsidize activities aiming at such reductions. Two options have been selected for further analysis:

- Direct subsidies to ‘Food Banks’ (enabling them to provide low-income households with food that would otherwise be wasted);
- Reduction or minimisation of food waste as an award criterion in public procurement contracts for catering (this can be seen as a subsidy because additional public expenditure may be involved by accepting a possibly higher contract price in return for food waste reduction).

7.5.1 Waste flow

Organic waste including food waste is separately collected in many municipalities in the Netherlands. There are different sources for the amount of organic waste: household waste, municipal waste and service sector waste. The total amount of organic waste includes garden waste as well. Table 7.1 shows the relevant contributors to organic waste. In the organic waste collected from households and the service sector, there will be substantial flows of food waste, although we do not have an indication of the size of the amount. In 2005, the total amount of organic waste separately collected amounted 2.8 Ktonnes.

⁴⁵ For details by country, see Appendix I.

Table 7.1 *Development of organic waste in the period 2000-2007*

	2000	2001	2002	2003	2004	2005	2006	2007
Household waste separately collected	1,457	1,404	1,406	1,340	1,407	1,362	1,296	1,317
Household waste: large-sized garden waste	359	355	397	377	397	406	407	452
Municipalities: public garden waste	578	575	649	631	644	656	639	689
Service sector	n.a.	n.a.	103	134	150	155	n.a.	n.a.
Rest organic waste	n.a.	n.a.	1,043	987	264	262	n.a.	n.a.
Total organic waste	3,560	n.a.	3,598	3,469	2,862	2,841	n.a.	n.a.

Source: SenterNovem (2006b); Milieu and Natuurcompendium. (n.a. = not available)

The total amount of organic waste collected from households amounts 1,362 Ktonnes in 2005, which is approximately 45% of the total amount of organic waste separately collected. In 2007, it was 1,317 Ktonnes. Next to food waste, organic waste includes garden waste, but these sub-flows cannot be disentangled. According to SenterNovem (2006a), the separately collected amount of waste is roughly half of the total organic waste generated by households. About 34% of the household rest waste (3,598 Ktonnes) is organic waste, i.e. 1,400 Ktonnes of organic waste in 2005.

In addition to the amount of organic waste collected from households, the service sector also discards organic waste. The total amount of organic waste from the service sector was 155 Ktonnes in 2005. More recent numbers are not available. The service sector includes a number of sub-sectors, which might be particularly interesting for the food banks to make arrangements with, like the whole sale companies (SBI74 codes 61 and 62) including auctions for agricultural products and flowers (SBI74 code 87) and the hotel and catering industry (SBI74 code 67). The amounts of organic waste of those subsectors are 31 and 36 Ktonnes respectively.

7.5.2 Analysis

Subsidizing food banks

Food banks are organisations that collect food from producers and retailers and distribute it among households that cannot afford it. The products concerned meet all quality standards for human consumption, but are for some reason not suitable for sale (e.g. due to a short period until the 'best before' date, or errors on the packaging). If the food would not be given to the food banks, it would normally be discarded as waste or processed as feed.

By the end of 2007, there were 105 food banks operating in the Netherlands, with some 13,000 clients (Arts, 2008). A recent 'quick scan' showed that among a sample of municipalities with a food bank on their territory (or in their region), in 47% there were contacts and/or agreements between the municipality and the food bank. Within this sample, in 40% of the cases (i.e. 19% of the total sample) the food bank received a

subsidy from the municipality (SGBO, 2008). The primary motives for subsidies to food banks are social ones. The total subsidy amount is unknown.

Given the fact that subsidies to food banks are currently provided by municipalities, it seems obvious that this level of public administration is also the most suitable one for providing environmentally motivated (additional) subsidies. However, it seems doubtful if municipalities would be prepared to increase their financial support to food banks, or, in case they do not subsidize them yet, to start doing so. There seems to be a general feeling among local policy makers that direct food support is not the 'first-best' option in poverty policy and that food banks should not be necessary in a social welfare state. In our view, this attitude is unlikely to change radically if waste policy motives are added to the social motives.

In any case, the extent to which food bank subsidies might contribute to less food waste is unlikely to be very significant. Tentatively, if we estimate the value of the food per 'client' of a food bank at EUR 500 per year, the total value for all present food banks would be around EUR 6.5 million. It seems unlikely that waste policy subsidies would more than double this amount, given the fact that other restrictions (such as availability of voluntary staff and of food meeting the standards) might become bottlenecks. As indicated above, the size of food waste is estimated at some EUR 1.5 to 3.5 billion per year, leading to the conclusion that the potential role of food banks in reducing this would probably be less than 1%.

Food waste reduction as a criterion in public tenders

The Dutch government aims at 100% sustainable public procurement by 2010. The implementation of this objective is supported by the 'Duurzaam Inkopen' programme. Under this programme, documents are published specifying the criteria for sustainable procurement in various product and service groups. One of these is the document specifying the criteria for catering (SenterNovem, 2008b). These criteria do not contain specific targets for the minimization of food waste. They only suggest including a provision in the contract with the catering company saying that a plan will be made to reduce its environmental impact, including, among others, food waste.

Food waste reduction could be given a stronger position in public procurement by specifying it as an award criterion. Tenderers could be given additional points in the award procedure if they succeed in reducing the amount of food waste below a certain reference level. In this way, the governmental organisation can 'buy' food waste reduction, since the additional points may outweigh a possibly higher price in the award procedure. The specification of the criterion (e.g. to express it in absolute (tonnes) or relative (percentages) terms; choice of the reference level; definition of food waste; etcetera) as well as the method of monitoring will of course need further elaboration.

A quantitative assessment of the possible effectiveness of this option is beyond the scope of the present study. This would require a detailed knowledge of the current situation regarding (avoidable) food waste in offices, the technical and economic potential to reduce these, and the costs of achieving this reduction. However, the size of the market involved is considerable. In 2007, the catering industry had a turnover of EUR 1.2

billion, and some 36% of the catering locations were government offices, educational, health and justice institutions⁴⁶ (their share in terms of turnover is unknown).

One might argue that the amount of food waste from catering services in public offices is to a large extent determined by the behaviour of the individuals using these services (canteen visitors etcetera), and therefore beyond the caterer's control. However, the caterer may find creative ways of offering the food and incentives to influence this behaviour, so as to ensure that the food he provides ends up in the clients' stomachs rather than in the garbage container. Including food waste reduction criteria in public tenders could mobilize this creativity.

The 'price' to be paid for the food waste reduction will not necessarily be a monetary one. Tenderers are likely to be able to achieve higher reduction rates if certain requirements in the tender, e.g. concerning the availability and diversity of food products and the ways of preparing, displaying and offering them, are made less stringent. For the canteen visitors this may imply that they may not always be able to get their favourite menus.

7.6 Case study VAT

Giving price incentives for environmentally desirable consumption using existing tax schemes has the advantage that administrative costs can be kept low, as the 'infrastructure' is already in place. Taxes on vehicles and fuels, for example, have been successfully differentiated to speed up the market penetration of catalytic converters and unleaded petrol.

The Value Added Tax (VAT) has also repeatedly been mentioned as a possible tool for stimulating 'cleaner' products. As VAT applies to the sales of almost all consumer goods and services, its scope is potentially very wide. Moreover, EU VAT legislation allows member States to apply one or two reduced rates (minimum 5%), in addition to the standard rate (minimum 15%).⁴⁷ In principle, this creates an opportunity for rate differentiation by bringing environmentally preferable goods under the low rate.

However, the current VAT Directive (2006/112/EC) restricts the application of reduced VAT rates to a limited number of goods and services. None of these are environmentally motivated, though the option of a reduced VAT rate for certain labour-intensive services (such as repairing bicycles, shoes and clothing) can be seen as an instrument that may help to extend the lifetime of products and thereby reduce waste. The Netherlands is one of the Member States that uses this opportunity.

The European Commission is currently reviewing the VAT system, including the issue of reduced rates. It has presented a Communication on the subject⁴⁸, and a proposal for some (minor) amendments⁴⁹. In the latter, the Commission announces that a more in-depth review of the whole structure of VAT reduced rates will follow.

⁴⁶ Source: Veneca.

⁴⁷ Presently, all Member States except Denmark apply one or two reduced rates.

⁴⁸ COM(2007) 380 final.

⁴⁹ COM(2008) 428 final.

The review of the system of reduced VAT rates might provide an opportunity to give it a 'greener' character, for instance by applying reduced rates to products or services that can contribute to waste reduction. As an example, the present chapter will discuss the option of applying a reduced VAT rate to the purchasing of second-hand goods.

7.6.1 Waste flow

Potentially, many different waste flows may be involved in this option. As the measure is intended to extend the life of durable goods, any kind of waste from these goods can be affected.

According to Statistics Netherlands (CBS), the number of firms in antiques and second-hand goods (SBI 5250) was 2,920 on 1 January 2004. CBS does not report on the turnover figures of this branche.

An obvious trend is the growth of direct trading in used goods between private households through the internet. The main Dutch website in this respect is 'marktplaats.nl', which contains some 6 million advertisements, attracts some 60 million visitors per month, and brings about a turnover of some €5 billion per year. Interestingly, the growth in internet-based trading has not led to a decrease in the turnover of 'recycle shops'.⁵⁰ This could be an indication that the total volume of trade in second-hand goods is increasing, and thus the amount of waste discarded decreasing, but we do not have quantitative data to confirm this.

7.6.2 Analysis

The existing VAT system already contains a specific provision, the so-called 'margin scheme', for used goods, art objects, collectors' items and antiquities.⁵¹ Re-sellers of these goods are allowed to account VAT over the profit margin instead of the full sales value of the good, provided that no VAT was charged when they purchased it (which is usually the case when buying from private persons). This provision is intended to prevent a accumulation of VAT on goods which enter the trade circuit for a second time.

To goods under the 'margin scheme' the usual VAT rate applies. For some products, for instance books, this is the reduced rate of (presently) 6%, but for many others, such as furniture, textiles and household appliances, it is the standard rate of 19%. The option that will be analyzed here is to apply the reduced VAT rate of 6% to all goods under the margin rule. Such a VAT reduction would enable second-hand traders to reduce the price of the goods they sell (and/or to pay a higher price for the goods they purchase)⁵², thus creating an additional stimulus for the trade in used goods.

Among the potential fiscal instruments to encourage second-hand trading, this is probably the most appropriate one. It will be very simple from an administrative point of view, because the activities to which the scheme would apply are already subject to a

⁵⁰ Source: Het Financieele Dagblad, 13 August 2008.

⁵¹ Wet op de Omzetbelasting 1968. Afdeling 5, artikel 28b – 28i.

⁵² Depending on market conditions, the measure could also lead to a higher profit margin for the trader.

separate VAT regime and therefore clearly distinguishable.⁵³ On the other hand, it would require a change in the current VAT Directive (2006/112/EC), for which unanimity in the European Council is required.

Presently, Belgium already applies a reduced VAT rate to so-called ‘recycle shops’, which provide employment to low-skilled unemployed people. This is made possible by Annex III of the VAT Directive which allows (under item 15) the application of the low VAT rate to the “supply of goods and services by organisations recognised as being devoted to social wellbeing by Member States and engaged in welfare or social security work”. A disadvantage of this option is that it may lead to unfair competition between ‘recycle shops’ and other (commercial) shops for second-hand goods.⁵⁴ Moreover, this measure would only apply to part of the market and therefore be less effective in terms of promoting recycling.

Quantitatively, little can be said about the expected effectiveness of the selected option. Data on the size of trade in second-hand goods are scarce, and data on the price elasticity of demand and supply of these goods are lacking altogether. However, some general considerations can be made:

- Used goods are increasingly being traded directly between private people, using the internet. This type of trade is obviously not affected by the VAT reduction.
- The magnitude of the price incentive from a reduced VAT rate is naturally limited to the difference with the standard rate (presently 13 percentage points in the Netherlands). For most second-hand goods, it is probably reasonable to assume that demand is price inelastic (i.e., a price elasticity of demand smaller than 1 in absolute value). The expected increase in demand is therefore likely to be less than 10%.
- For some types of goods (e.g. cars, refrigerators) a substantial part of second-hand trading takes place with foreign countries. To the extent that VAT reduction would increase the demand for second-hand products in the Netherlands, it may imply a decrease in the export of these products. The overall impact (in terms of global average life extension within the product group) will then depend on the way in which the demand abroad is met that was previously covered by goods from the Netherlands.
- A growth in the use of used goods (instead of new goods) does not necessarily have positive net environmental benefits. Though it may reduce the amount of waste, it may also increase energy use (in the case of e.g. household appliances that remain in use for a longer time, slowing down the market penetration of newer, more energy efficient models).

⁵³ The application of the reduced VAT rate could also easily be restricted to ‘used goods’ (which are defined in the ‘Wet op de omzetbelasting’ 1968, art. 2a, 11), leaving art objects, collectors’ items and antiques under the standard rate.

⁵⁴ See article in NRC Handelsblad, 27 May 2008.

7.7 Conclusions

7.7.1 Subsidies

Even though subsidies seem to be in conflict with the ‘polluter pays principle’ they can still play a useful role in waste policy, either because of positive externalities (e.g. innovation) or because the ‘first best’ option of taxing the negative externalities is not feasible.

Subsidies and preferential public procurement can play a role as waste policy instruments in situations where a strict application of the ‘polluter pays principle’ is not feasible, e.g. for practical or political reasons. In this chapter, we have looked at two options in the area of food waste. Subsidizing food banks does not seem to be a very effective way of preventing food waste, and will in any case only play a marginal role. The application of food waste reduction as an award criterion in public tenders, on the other hand, might be a more substantial option and in our view deserves attention within the framework of the present ‘Sustainable Procurement’ campaign.

7.7.2 VAT

A reduced VAT rate for trading in second-hand goods under the ‘margin scheme’ gives a financial stimulus for this type of trading by reducing the amount of VAT to be paid by 13% of the net selling price (at current VAT rates in the Netherlands). This measure can be introduced at low administrative cost, but requires amendments to EU VAT legislation. Quantitatively, little can be said about the impact, due to a lack of data, but it is unlikely that the demand for used goods will grow by more than 10%. Furthermore, the effectiveness of this option will be reduced by the ongoing trend towards direct second-hand trading between households, and its environmental merit is uncertain, especially for energy using products.

VAT reduction might be more appropriate for products and services that are ‘intrinsically’ less waste-intensive. (i.e., by their nature lead to lower amounts of waste), especially in situations where there is a close substitute that is ‘waste-intensive’ (which would remain under the high VAT rate). One could, for example, think of ‘real nappies’ (including the associated laundry service) as opposed to disposable ones. Nevertheless, the limitations of the VAT instrument (especially the limited price difference that it can bring about) should be kept in mind, as well as the political effort required to achieve the necessary unanimous support from EU Member States.

8. Synthesis and recommendations

8.1 Introduction

Economic instruments gain popularity among policy makers who are, among others, responsible for designing waste policies. The attractiveness of economic instruments lies in their capacity of encouraging households and producers to improve their waste management behaviour. In addition, taxes and charges are able to generate funding for waste management activities. Despite the increasing application of economic instruments in waste management, the capacity of these instruments to deliver efficient solutions for waste problems may not yet be fully exploited.

This study aims to support Dutch and European policy makers in their decisions on economic instruments in waste management. The overall goal of the study is to explore the opportunities for extended use of economic instruments for waste policy in the Netherlands. This was achieved by conducting an inventory of the current practice of economic instruments in waste management in the Netherlands and abroad and by analysing the feasibility of a number of economic instruments for a range of waste materials. In the following, a summary is provided of the main findings and recommendations are formulated for policy makers in the Netherlands and other countries involved in designing waste policies.

8.2 Summary

Waste tax

In theory, taxes on final waste disposal are the most efficient policy instruments to internalise the environmental externalities caused by waste. They leave the choice between final disposal and other options to the market, thus ensuring that for each batch of waste the option with the lowest social cost will be chosen. In a well functioning market, the price signal from the waste tax will be transferred along the value/material chain, providing appropriate incentives to all actors to look for alternatives for final disposal, such as prevention, material substitution and recycling.

In practice, however, waste taxes are rarely based on environmental externalities and do not differentiate between waste streams and processing methods according to their environmental impact. Moreover, due to imperfect waste markets, the price signal created by a waste tax rarely reaches the household or firm that takes the waste disposal decision. Nevertheless, even these ‘imperfect’ waste taxes can play a role in waste policy by making the landfilling and/or incineration option financially less attractive than other options that have a higher rank in the waste hierarchy.

Several countries in the EU and elsewhere apply waste taxes, most of which are restricted to landfilling. Differentiations in landfill tax rates between types of waste are common (e.g. combustible versus non-combustible waste). Some countries apply lower rates to landfill sites that meet certain environmental criteria. Similar differentiation patterns are found for incineration. Generally, revenues from waste taxes accrue to the general budget, but several countries earmark them for specific waste related or other

environmental purposes. Because waste taxes are a part of a waste policy package it is difficult to disentangle their impact from the effect of other instruments. Some evidence is found on landfill taxes in Austria, UK and Denmark suggesting that these taxes led to significant reductions in the landfilling of certain waste types or promoted the use of landfill sites that meet certain environmental standards.

The case study demonstrated that introducing (higher) taxes on incineration and landfilling without other instruments is hardly effective. An incineration tax will raise the price of incineration over the price of landfilling, making it more attractive to landfill waste. Combining a landfill and incineration tax can potentially stimulate recycling if these higher costs are passed on to the households. However, since this is usually not the case, it doesn't lead to more recycling. The service sector is somewhat more sensitive for changes in the waste tax, but the high costs of recycling prevent them from switching in a notable manner. Likewise, municipalities are constrained in their flexibility to switch given the lack of unutilized capacity and the long-term contracts with waste incinerators.

Waste collection charges

The literature recommends setting the price that is to be charged to households in a system of differential and variable rates (DVR) equal to the marginal cost of providing the service. This provides an incentive to households not to over consume the service while forcing bureaucrats to maintain greater cost transparency. A social cost-benefit estimate for DVR systems shows that by following this principle, the net social benefit of DVR systems is mostly positive. The literature also reveals that weight-based and pay-per-bag schemes perform best, while schemes for which the charge is based upon the volume of the container perform worst. A precondition for good performing DVR schemes is the presence of a comprehensive system for collection of segregated materials for recycling. According to the literature, the effects of DVR on illegal waste disposal remain difficult to estimate.

Local authorities in several countries apply DVR systems to some degree. In Korea, for example, a mandatory 'pay-per-bag' scheme for all municipalities is applied nationwide, with the municipalities deciding on the specifics of the bag and its price. The revenues are supposed to cover the costs of accompanying (free) collection services for recyclable and compostable waste. The Korean scheme has led to substantial growth in recycling activities and in waste prevention initiatives. After the initial stages, illegal disposal was not found to be much of a problem. In Europe, no country has yet taken the step towards mandatory DVR although France, Ireland and Italy have expressed their intention to do so. On the other hand, some European countries (such as the UK and Greece) do not even allow their municipalities to apply DVR.

The analysis confirms DVR to be an effective instrument in reducing the supply of unsorted household waste while promoting recycling. Many of the often-cited objections against DVR appear to be invalid: DVR can be successfully applied in large cities without increasing the costs of waste collection or leading to more illegal disposal. In fact, the total amount of household waste would decrease with the implementation of DVR while increasing the separation of waste by households.

In short, the use of DVR in waste collection charging can be seen as a necessary complement to waste taxes. A DVR system ensures that the price incentive of the waste

tax is transmitted to households and thus contributes to waste reduction and recycling. In turn, DVR itself needs complementary measures (such as adequate facilities for separate collection) to be fully effective.

Taxes on raw materials and products

In theory, taxes on raw materials and products are not considered effective tools for addressing problems in the waste stage. Economists generally argue that policy instruments should directly intervene at market failures: if the market failure exists at the end of the chain (in the waste stage), imposing a tax at the beginning of the chain (raw materials) would therefore not be appropriate. On other hand, the literature does support the use of virgin material taxes in the context of more general policy objectives such as promoting recycling, of reducing material intensity in production and consumption, or discouraging the use of specific materials in particular.

For a product tax to be an effective environmental policy instrument, it will need to address the environmentally undesirable properties of the product as directly as possible. For example, a general tax on packaging may be justified if the primary objective is to reduce resource use and waste in general, but it may need to be differentiated or confined to specific materials if it is intended, for instance, to reduce the amount of non-biodegradable waste. Moreover, specific circumstances may call for a product tax to be embedded in a package of instruments, including non-economic ones. For example, the literature shows that a tax on a consumption good, in combination with a recycling standard, a material subsidy and a subsidy on recycling services, might be an appropriate policy instrument under conditions of imperfect competition in the recycling industry. Moreover, to limit transaction costs, product taxes are most suitable for products that are consumed in large numbers and/or with a small number of producers and importers.

Several EU countries apply taxes on the extraction of natural resources. A study assessing the impact of taxes and charges on sand, gravel and rock in four EU Member States clearly shows that countries with relatively high tax/charge rates also have relatively high recycling rates. However, the study emphasizes that in all cases the tax/charge is just one factor among many, including other policies, technical factors, and specific

national circumstances. Generally, taxes on raw materials do not seem to have much impact in terms of a reduction in the demand for virgin materials and a shift towards recycled materials. Moreover, taxes on raw materials may have unintended side effects such as an increase in energy consumption or a shift from local to imported materials.

The analysis confirms that taxes on primary raw materials can contribute to a reduction in the use of such materials and to a better competitive position of recycled or renewable alternatives. However, the virgin material and product tax would be far more effective if it would be accompanied by measures to increase waste separation by the service sector and the households. After all, if the supply of recyclable waste does not increase it will be difficult to switch from virgin material to recycled material. Also, the design of the taxes should explicitly aim at minimizing possible unintended side effects.

Deposit refund schemes

The literature is early to recognize the advantages of deposit refund schemes (DRS). By reducing illegal dumping, DRS greatly reduces monitoring costs. Moreover, DRS can provide firms with incentives to prevent losses of the material. The literature also defines the conditions under which DRS can be considered an effective policy instrument. Some scientists even claim that DRS should be applied to all products in combination with a waste tax.

In practice DRS is mainly applied for beverage containers. In most countries DRS is implemented on a voluntary basis. Occasionally, DRS is stimulated by allowing successful companies to enjoy privileges, e.g. a reduced rate or exemption from packaging taxes. A number of countries have been applying mandatory DRS. The types of beverage containers covered differ (e.g. they may or may not include non-refillable containers such as cans), as do the deposit rates. Generally, these systems lead to high return rates and a reduction of littering. On the other hand, the handling and administration costs can be substantial.

The analysis shows that DRS can be effective in redirecting waste streams from final disposal to reuse and recycling. However, a wide collection network is a prerequisite for the success of DRS. In the cases of batteries and small white appliances, there is presently a wide network of collection that resulted from established producer responsibility schemes. In the case of batteries, the current levels of separate collection (on a voluntary basis) compared to the level of disposal are already relatively high. Therefore, a DRS will only modestly contribute to more separate collection of batteries, while the contribution in the case of small white appliances will be more significant.

VAT reduction, subsidies and public procurement

Subsidies and tax benefits do not adhere to the 'polluter pays principle' and are generally not the most efficient economic instruments to address waste externalities. On the other hand, they may still be useful policy instruments in situations where, for whatever reason, waste taxes cannot be applied. Subsidies on reuse and recycling are particularly promising as part of an instrument package. The well-known disadvantage of subsidies of high administrative costs can be overcome by linking subsidies to (fiscal) schemes that already exist (e.g. the VAT or taxes on income and profit). In reality, financial support from government budgets to activities that contribute to better waste management is given in a variety of ways across countries. Examples include direct grants (for R&D or investments), conditional money transfers to lower authorities, 'soft' loans, and tax reductions.

The qualitative analysis showed that subsidies and preferential public procurement (which can also be seen as a kind of subsidy) can play a role as waste policy instruments in situations where a strict application of the 'polluter pays principle' is not feasible, e.g. for practical or political reasons. In the food-related cases considered here, mixed results emerged. On the one hand, subsidizing food banks does not seem to be an effective way of preventing food waste. On the other hand, the application of an award criterion on food waste reduction in public tenders is more promising and thus deserves attention within the framework of the present 'Sustainable Procurement' campaign in the Netherlands.

The second case study considered dealt with a reduced VAT rate for trading in second-hand goods. Although this measure can be introduced at low administrative cost, it would require amendments to EU VAT legislation for its implementation. Lack of data makes it difficult to make strong predictions on the impact of a lower VAT rate. However, due to the limited price difference that it can bring about we do not expect the impact to exceed 10% increase in trade volume. Moreover, the effectiveness of this measure is further reduced by the growing trend of direct trade of second-hand between households via the Internet. In general, VAT reduction seems to be more suitable for products that are ‘intrinsically’ less waste-intensive.

8.3 Conclusions and recommendations

By combining a literature review, an inventory of foreign experiences and economic analysis, this study shows that economic instruments have an important role to play in waste management. The case studies presented in this report mainly illustrate the underlying mechanisms and possible impacts of economic instruments in waste policy, applied to a selective number of product-waste chains. Any generalisations should therefore be treated with caution.

Despite this reservation, it can be concluded that there is scope for a wider and more intense use of economic instruments in waste policy in the Netherlands. In theory, economic instruments generate strong incentives to make people behave in a more environmentally friendly way, and to move waste streams to higher levels in the waste hierarchy. Dutch waste policy is currently characterised by a mixture of direct regulation, economic instruments and other instruments (such as voluntary agreements and information provision). This study demonstrates that there is a clear potential for expanding application of economic instruments in the Netherlands.

However, the potential of economic instruments is limited by a number of ‘real world’ conditions. First, price incentives are often not reaching the actors who have to change their behaviour. This is partly caused by inefficient design of economic instruments, for instance, in the case where price signals intervene too early in the product chain to have an effect on the waste stage. Otherwise, price incentives may be distorted due to the absence of a mechanism to transfer incentives to households and the service sector. This is the case for the waste tax in the Netherlands, which is not always effectively passed on to the waste generators. Second, in many cases recycling is still more costly than the conventional way of managing waste. Therefore, making recycling more attractive may sometimes require substantial political support to release additional public funds. This support is not always there. Finally, economic instruments never operate in a closed system. Even at a local scale, leakage of effects may occur, for instance in the case of the increase of illegal dumping as a result of the DVR schemes. At a national and international scale, such leakage effect also may occur (e.g. waste export).

Policy makers should take into account these potential barriers in designing economic instruments. For example, decision makers should take into account the lesson that the instruments should intervene as close to the subject at stake as possible to prevent distortions avoiding the instrument to be effective. Also, policy makers should explicitly map out the existing market imperfections before implementing new economic

instruments. In fact, it may sometimes be more effective to change the conditions in which waste policies are operating, rather than add more policies. Nevertheless, it will be inevitable for policy makers to sometimes accept second-best choices, while trying to limit the negative consequences as much as possible by explicitly addressing negative side effects. Being part of the EU, the freedom of the Dutch government to implement new economic policies is limited.

One way of following a rational and realistic approach in expanding the influence of economic instruments in waste policies in the Netherlands is by explicitly addressing their potential effectiveness and feasibility as well as the conditions that need to be fulfilled and the potential obstacles and limitations. For the instrument categories that have been considered in this study, the main issues can be summarized as follows.

Waste taxes

An increase in the tax on landfilling and the introduction of a non-zero tax rate on incineration will only be effective if simultaneous measures are taken which transfer the incentives of the tax on to the producers of waste (i.e. households and the service sector). This can be achieved, for example, by introducing unit-based pricing (DVR; see below) and stimulating separate collection of waste streams. Moreover, alternative treatment capacity should be sufficient to allow for switching to the most desirable waste treatment option. There should also be sufficient confidence that the lowest-cost alternatives (including those abroad) are indeed the environmentally preferable ones. Further research is needed to explore to what extent these conditions are met or can be met in the near future.

Waste collection charges

DVR has proven to be (under certain conditions) a cost-effective instrument for waste reduction and more recycling. Moreover, DVR can itself be seen as a necessary condition for the successful implementation of other instruments, such as waste taxes (see above). It is therefore recommended to stimulate municipalities to adopt DVR schemes, accompanied by the provision of adequate facilities for waste separation by households. Before introducing DVR in 'very strongly urbanized' municipalities (i.e. the 12 largest cities in the Netherlands) a pilot experiment in one of them might be useful to test the behavioural response in this category.

Taxes on raw materials and products

The introduction of new taxes on specific raw materials and products, and the increase in the tax rate for existing ones (such as the packaging tax) can in principle contribute to higher recycling rates and a lower resource intensity in general. However, the presence of other incentives to increase the supply of recycled materials is essential for the effectiveness of such taxes. Moreover, (very) high tax rates will be needed to achieve any significant impact, and such high rates may cause unwanted side effects (evasion; substitution by untaxed, but equally undesirable alternatives). Revenue raising rather than waste reduction might therefore be the main consideration justifying the use of such taxes (at relatively low rates).

Deposit refund systems

Deposit refund systems (DRS) have a strong potential to achieve high collection and recycling rates. Their application therefore deserves consideration, especially with respect to (discarded) products for which separate collection rates are currently low, while an adequate infrastructure is already in place or can be created without excessive cost. Small electric appliances may be a case in point. Before introducing (mandatory) DRS, their impact on international trade and interactions with other policies (e.g. in the framework of the WEEE Directive) should be further investigated. The administrative costs and overall environmental impact would also require more detailed analysis.

VAT reduction, subsidies and public procurement

Positive financial incentives for ‘low-waste’ products and services can be justifiable exemptions to the ‘polluter pays principle’ under certain circumstances. They can be given in a number of different ways. Among the examples analysed in this study, the inclusion of waste reduction clauses in public tenders has been identified as a promising candidate. The potential effectiveness and feasibility of subsidies and other positive incentives should be analysed on a case-by-case basis. Their introduction may depend on consent at EU level (fiscal harmonisation; state aid rules).

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Appendix I. Economic instruments abroad

In this appendix, a brief survey is presented of economic instruments used in other countries, both in the EU and elsewhere. Completeness is not aimed at; instead, the survey should provide useful examples and insights for innovations in the use of economic instruments in the Netherlands. The focus is therefore on instruments that differ from those applied in the Netherlands, for example because they relate to other products and waste streams, or because they have a different structure, higher tax rates, etcetera.

I.1 Waste taxes

I.1.1 Introduction

A growing number of countries apply a tax on the final disposal of waste. The present chapter only deals with examples from countries where a waste tax scheme is used that clearly differs from the Dutch situation, e.g. because it applies not only to landfilling but also to incineration, or because it differentiates between types of waste or between landfill features.

I.1.2 Austria

In Austria, since 1996 landfill tax rates are differentiated according to the technical quality of the landfill site and to the type of waste. Since 2006, the basic rate⁵⁵ is EUR 87 per tonne. A surcharge of EUR 29 per tonne applies to landfills that have no basement seal system or no vertical enclosure. Another surcharge of EUR 29 per tonne applies to municipal waste landfills that are operated without a landfill gas collection and treatment system (EIONET).

According to the Austrian Federal Environment Agency, this differentiation has been a clear incentive to modernise the Austrian landfills: whereas in 1996/97 21 sites did not meet the latest technological standards, in 1999 this was true for only 4 sites (Umwelbundesamt, 2000; cited in Bartelings *et al.*, 2005).

I.1.3 Belgium

The region of Flanders used to have a complicated system of waste taxes, but since 1.1.2007 this has been simplified. There is now a single rate for incineration (EUR 7 per tonne) and there are two rates for landfilling: EUR 75 for combustible and EUR 40 for non-combustible waste (MIRA).

The region of Wallonia levies a waste tax on communities where the average amount of municipal waste is more than 240 kg per inhabitant per year. The amount of waste that exceeds this threshold is taxed at a rate of EUR 35 per tonne (Fiscalité Wallonie). Reusable waste that is collected separately is not subject to the tax, in order to support

⁵⁵ Lower rates apply to certain categories such as construction and demolition waste.

waste recycling. Moreover, any material that still could be eliminated, segregated, or recycled from the household waste after the collection is not taken into account. In this context, recycling is also understood to include thermal recovery. In essence, therefore, the tax can be considered to be a landfill tax (Eunomia, 2002).

1.1.4 Denmark

The Danish tax on waste, introduced in 1987, covers both incineration and landfilling. The purpose is to provide an incentive for waste generated to be directly recycled or exploited by incineration with energy recovery, instead of being landfilled. Moreover, the purpose is to prevent the generation of waste (EIONET).

Since 1993, the rate for incineration is set at a somewhat lower level than the rate for landfilling. In the years 2003-2005, incineration was taxed at DKK 330 (EUR 44) per tonne and landfilling at DKK 375 (EUR 50). In the period 1997-2001, a lower rate was applied to incineration with energy recovery than for incineration without energy recovery. Revenues from the waste tax were around DKK 1 billion (EUR 130 million) in recent years (Speck *et al.*, 2006).

The Danish waste tax has led to some reduction in the amount of waste for final disposal, especially in the heavier components of household waste and in construction waste and mixed waste. The differentiation between landfilling and incineration has led to some diversion towards incineration, as was intended (Speck *et al.*, 2006).

The tax was not meant to provide an incentive for individual households, but to make it more profitable for the refuse collection authorities in the municipalities to establish recycling and sorting systems. Andersen (1998) notes that it was implicitly assumed that the tax would induce municipal authorities to minimize costs rather than simply passing the tax on to residents.

1.1.5 Estonia

Estonia has a fairly comprehensive system of waste-disposal charges. The charge rates are differentiated between different types of wastes and also vary depending on the location of the landfill site and whether the site meets predefined environmental standards (EEA, 2005b, p. 64).

1.1.6 France

Since 2003, the French landfill tax has two different rates for non-hazardous waste, depending on the presence of an environmental management system in the landfill company. As of 1 July 2007, the rate for landfills with EMAS or ISO 14000 certification was EUR 8.10 per tonne. For other landfills it was EUR 9.90 per tonne (CEWEP).

The 'Round Table' on waste under the French 'Grenelle Environnement' has recently proposed to increase the rate of the landfill tax to EUR 20-40 per tonne and to introduce a tax on waste incineration of EUR 5-10 per tonne, the rate depending on energetic efficiency (Grenelle Environnement, 2007).

1.1.7 Ireland

Ireland has a landfill tax of EUR 15 per tonne since 2002. The revenues are ring-fenced to support waste minimisation and recycling initiatives (EIONET).

1.1.8 Italy

Italy applies a landfill tax since 1996. The rate is differentiated to take into account local conditions and environmental costs (EEA, 2005a). For municipal solid waste, the rate varies between EUR 10.33 and 25.82 per tonne. The tax also applies to incineration without energy recovery, which is taxed at 20% of the standard rate. The purpose of the tax is to give incentives to prevention and recycling. The tax revenue can be used by the local administration in order to finance environmental projects (EIONET).

1.1.9 Latvia

The Latvian landfill tax differs according to the degree of hazard associated with a particular waste. Since 2002, municipal waste is taxed at a relatively low rate (EUR 1.07 per tonne, or, if no weighbridge is present, EUR 0.34 per m³). Tax rates for hazardous and highly hazardous waste amount to EUR 14.29 and 71.43 per tonne, respectively (Speck *et al.*, 2006).

1.1.10 Norway

The waste tax in Norway applies to both landfill and incineration and was introduced in 1999 in order to help reduce the amounts of waste produced. The waste taxation system was changed in 2003. The changes were to be revenue neutral and were not designed to affect the relative status of landfill versus incineration (Speck *et al.*, 2006).

Since July 2003, landfill tax rates have been differentiated according to the environmental standard of the landfill site to which the waste is delivered. The higher rate (NOK 533 or EUR 65 in 2005) applies to sites that do not fulfil requirements with regard to site linings. The standard rate was NOK 409 (EUR 50) per tonne (Speck *et al.*, 2006).

With regard to incineration, until July 2003, a standard rate applied in the case of 100 percent energy recovery and a maximum additional rate applied in the case where no energy was recovered. The standard plus the maximum additional rate amounted to the same rate as that for landfill. Reductions towards the standard rate applied as the percentage of energy recovery increased. Since 2003, waste delivered for incineration has no longer been differentiated according to the degree of energy recovery, but rather the environmental quality of the waste delivered to waste treatment plants, i.e. the tax now relates to the pollutant content of the emissions from burning the waste. The aim is to better reflect the environmental costs of incineration as well as to stimulate energy recovery from waste incineration. Furthermore, a CO₂ tax is now charged per tonne of waste delivered to incineration plants. Plants burning waste that does not include fossil material are excluded from the charge (Speck *et al.*, 2006).

The revenues in 2005 from the Norwegian waste tax were estimated at NOK 720 million (EUR 91 million) (Speck *et al.*, 2006). Since the introduction of the tax, the share of

landfilling of household waste has dropped from 43% to 24%, while recycling and incineration have increased from 33% to 45% and from 23% to 31% respectively (OECD, 2004a). No information is as yet available on the impact of the changes in the tax structure since 2003.

1.1.11 Poland

Poland has a landfill tax with different rates for various waste types that are collected separately (OECD/EEA database). The rates for municipal waste ranged from PLN 13.8 (EUR 4.00) to PLN 44.3 (EUR 12.85) per tonne in 2001 (OECD, 2004c).

1.1.12 Slovakia

Slovakia has a scheme in which municipalities will pay a levy on landfill to the local authority in whose borders the landfill is located. The level of payment is related to the number of fractions being separately collected by the municipality sending the waste for disposal. A similar waste charge is also in place in the Czech Republic (EEA, 2005b, p. 64).

1.1.13 Spain

The region of Catalonia introduced a landfill tax of EUR 10 per tonne in 2003 (OECD, 2005a). The revenues are used to support the development of schemes for separation of biowastes at source, to be implemented by municipalities. Municipalities receive support based on a per tonne basis, which varies with the level of contamination of the collected material (EEA, 2005b, p. 64).

1.1.14 Sweden

Under the Landfill Tax Act, in force since January 2000, all material entering landfill facilities is taxed, while material removed from the facility qualifies for a deduction. The current rate is SEK 435 (EUR 47) per tonne. The impact of the tax in terms of increased recycling and reduced landfilling has been considerable.

On 1 July 2006 a tax was imposed on the fossil component of waste disposed of by incineration. The aim is to encourage recycling (of plastics in particular), reduce carbon dioxide emissions, and encourage cogeneration (combined heat and power generation). The tax is levied on domestic waste only.

The amount of tax charged is based on an assumed fossil content, currently set to 12.6 percent of the waste stream. Waste incineration facilities with only heat generation pay SEK 444 (EUR 48) per tonne of waste, while facilities that generate power as well pay a lower rate (Swedish EPA).

According to Sahlin *et al.* (2007), the main effect of the incineration tax will be an increase in biological treatment. The tax is not expected to contribute significantly to more recycling of plastic packaging waste.

1.1.15 United Kingdom

The impacts of the UK's Landfill Tax have been studied relatively intensively (see Bartelings *et al.*, 2005, section 2.3.3). Generally speaking, it has led to a substantial reduction in the landfilling of construction and demolition waste, while having hardly any impact on other waste types. Martin and Scott (2003), for example, conclude that "the tax has failed to significantly change the behaviour of domestic waste producers and SMEs". One of the reasons for its lack of effectiveness is the fact that the price incentive does not reach the households, as municipalities are not allowed to levy specific waste collection charges (see section 1.2.6).

1.1.16 Japan

In Japan there is a levy on disposal of recyclable materials. The levy is calculated using a scaling factor for the particular industry involved and a unit cost for the type of recyclable (MMA and BDA Group, 2003).

1.2 Waste collection charges

1.2.1 Belgium

Municipalities in Flanders have a great deal of autonomy in determining their waste policies. Currently, almost all municipalities apply a kind of differentiated waste collection charges. In 2003, the average charge rate was EUR 1.14 per bag (of 60 litres) for residual and organic waste. Municipalities sell blue bags for small packaging waste (plastic bottles, metals, cardboard beverage boxes) at a much cheaper rate (EUR 0.125 to 0.25 per bag), to encourage separate collection (OECD, 2007a). Between 1991 and 2002, the share of separately collected household waste in Flanders increased from 18% to 70% (MIRA).

In Wallonia, 70% of the municipalities apply a 'charge per bag' system for waste collection (OECD, 2007a).

Gellynck and Verhelst (2007) analysed the impact of different waste policy instruments. They could not find a significant impact of weight-based fees on the amount of waste collected (only 17 of the 295 municipalities applied a weight-based system). However, the level of unit-pricing by the bag appeared to have a significant impact (implicit price elasticity: -0.139).

Eunomia (2002) found that in the province of Brabant, where the cost of a waste bag were higher than elsewhere in Flanders, the average decrease in the amount of residual household offered was much greater than in the other provinces.

1.2.2 France

The 'Round Table' on waste under the French 'Grenelle Environnement' has recently proposed an obligation for French municipalities to apply waste collection charges with a variable as well as a fixed part (Grenelle Environnement, 2007).

1.2.3 Ireland

According to EEA (2005b, Box 3.2), Ireland has the aim to make differentiated user charges for waste collection compulsory across all municipalities.

1.2.4 Italy

Decree 22/97 (art. 49) radically modified the pre-existing tax on solid urban waste. According to decree 22/97, this tax will be gradually replaced by the Waste Tariff. The new Municipal Waste Tariff should be proportionate to the number of persons *and to the quantity of waste actually produced* (EIONET; emphasis added). In EEA (2005b, Box 3.2), such a nationwide ‘diftar’ policy for Italy is still referred to as being planned.

Eunomia (2002) reports on two local ‘diftar’ schemes in Italy.

1.2.5 Luxembourg

Eunomia (2002) reports on a pilot project with volume and weight-based charging in two communities. The scheme included different rates for different waste streams. Compostable waste was charged at EUR 0.09 per kg, so as to encourage home composting.

1.2.6 United Kingdom

The UK is characterised by the *absence* of a policy instrument in this area: local authorities are not allowed to apply waste collection charges for households that vary with the amount of waste set out for collection.⁵⁶ The costs of waste collection services are to be covered by the revenues raised through the so-called Council Tax, and other sources of funds for local administrations. Hence, the incentives for waste reduction and recycling that could be provided by the landfill tax are not passed on to those that generate waste in the first place. Braathen (2007) considers this to be a likely explanation for the little impact that the UK’s landfill tax has had.

1.2.7 South Korea

South Korea implemented compulsory nationwide DVR (differential and variable rate) charging for household waste in 1995. The system was established as a means to generate an economic incentive for waste reduction and to generate revenue to support a recycling service that is free at the point of delivery. The target wastes are municipal solid wastes from households, commercial sectors, small businesses and office buildings (those generating less than 300 kg per day). The system is accompanied by the provision of an increasingly comprehensive service for collecting recyclables and compostables.

A ‘pay-per-bag’ system was chosen since it was believed to be awkward to make use of rigid containers in such a densely populated country, and because the use of pre-paid sacks made revenue collection more straightforward. Each municipality is free to choose

⁵⁶ A similar prohibition on differential charging exists in Greece, where waste charges are collected through electricity bills according to the surface area of the served property (Karagiannidis *et al.*, 2006).

the specifications of the designated sacks, taking into account specific needs. The price of a bag should cover the costs of collection, transport and treatment as well as the costs of making the bag. In practice, bag prices vary greatly among local governments (e.g. between USD 0.22 and USD 0.70 for a 20-litre bag). Despite scale economies in waste management, bag prices are generally higher in metropolitan areas than in provinces. For recyclable municipal waste (paper, cans, bottles, metal and plastics), residents do not pay for waste services. For bulky waste, user charges also apply: a sticker must be purchased at a price set by the municipality according to the type and size of the item (e.g. furniture, white goods) (OECD, 2006c).

The scheme has contributed to a reduction in waste arisings of 15% and an increase in recycling from approximately 15% in 1994 to nearly 50% in 2004 (see figure I.1). Illegal disposal was found to be a problem following introduction in 1995 but an effective enforcement programme has reduced illegal disposal incidents to 13% of their 1995 levels (OECD, 2006a).

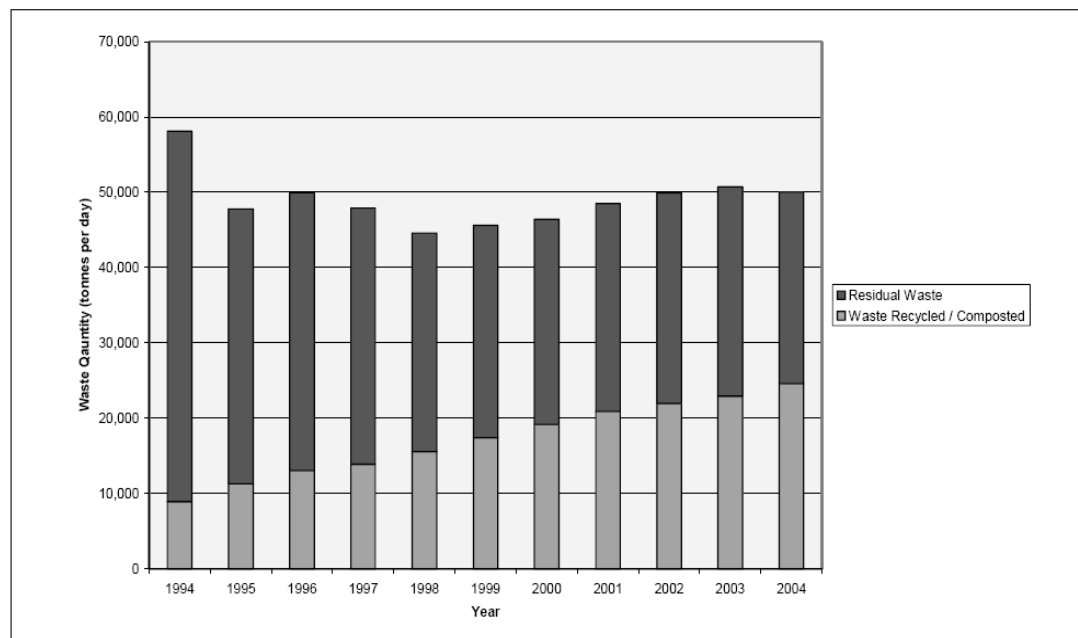


Figure I.1. *Effects of volume-based waste fee in South Korea (source: OECD, 2006a; original source: Korea Environmental Policy Bulletin, Update Version of Issue 1, Volume 1, January 2006).*

According to the South Korean Ministry of Environment, as a result of the the nationwide DVR the recycling-related industrial sector saw a rise in the number of recycling businesses and a sharpened competitiveness of re-producing companies in the market. Manufacturing and distribution industries also converted their production and sales procedures to curb excessive packaging and waste generation. Furthermore, notable changes have been witnessed in people's lifestyle as well. It is now common to see people swapping or purchasing second-hand goods, buying refillable goods that tend to produce less waste, and carrying reusable shopping bags (Ministry of Environment, ROK).

1.2.8 United States

Communities throughout the United States have traditionally levied fixed collection fees for household waste, or they have included collection and disposal costs in property taxes. However, a growing number of communities are now charging for solid waste collection based on the volume generated by the household ('pay-as-you-throw', PAYT). By 2001, such PAYT programs had been implemented in more than 4,100 communities in 42 states, reaching an estimated 10% of the U.S. population. Four states (Washington, Iowa, Wisconsin and Minnesota) had mandated the use of variable rate programs in some or all of their municipalities. In most areas where variable rate programs have been introduced, the amounts of waste collected have decreased significantly, a result of either increased recycling or decreased waste generation (EPA, 2001).

1.3 Taxes on raw materials

1.3.1 Belgium

Flanders has a charge on gravel extraction, amounting (in 2005) to EUR 0.39 or 0.56 per m³, depending on the location of the quarry (mountain or valley, respectively). Revenues (EUR 2.3 million in 2005) go to a 'gravel fund' (OECD, 2007a).

1.3.2 Bulgaria

According to EEA (2005b, Box 3.2), Bulgaria has a tax on aggregates.

1.3.3 Czech Republic

The Czech Republic applies a fee on the exploitation of mineral resources, amounting to up to 10% of the value of the extracted raw material. The revenues (75% for the municipality, 25% for the State budget), are to be used for remediating environmental damage caused by mining (OECD, 2006d).

1.3.4 Denmark

Since 1990 a tax levied on the extraction of raw materials has been in place in Denmark. The tax rate, fixed at DKK 5 (EUR 0.67) per m³ of raw material extracted, has not been revised since its introduction. In recent years, the tax revenues are about DKK 150 million (EUR 20 million) per year (Speck *et al.*, 2006). The raw material tax was introduced in close junction with the Danish waste (disposal) tax (see section I.1.4 in this appendix). The main intention of the two taxes in combination is to reduce the use of the above resources and encourage substitution of recycled materials (e.g. construction waste).

According to Söderholm (2006), it is difficult to evaluate to what extent the Danish tax on raw materials has lowered consumption of virgin materials and encouraged substitution to recycled materials. When introducing the tax in 1990 the Danish government expected the effects on consumption to be modest. The tax burden is primarily transferred from producers to end consumers with prices increasing between 3 and 33 percent depending on material. Tax costs for these consumers (primarily

construction and infrastructure companies) are however small in relative values, and their raw material demand is highly price inelastic. Extraction levels decreased between 1989 and 1993, but then increased again during the latter half of the 1990s. After 1999 the trend was downwards again. Overall, however, total extraction in 2002 was more or less the same as in 1990 when the tax was introduced. This confirms the conclusion made in Ecotec (2001) that the tax has had little effect on raw material extraction in Denmark. The derived demand for raw materials (in construction activities) combined with a low price elasticity of demand most likely explains this meager effect, but also implies that the annual rate of extraction represents a poor proxy for the effectiveness of the tax.

The effects of the tax are thought to be modest also in terms of encouraging the share of recycled materials, again since the tax rate is relatively low and since some raw materials may be difficult to substitute away from (partly because the demand for supreme quality is high in construction work where virgin raw materials serve as inputs). If there is any effect on recycling behaviour, it is instead likely to be related to the Danish tax on waste disposal. Ecotec (2001) notes that the disposal tax provides a much stronger economic incentive than does the tax on materials. As much as 90 percent of all demolition material is today recycled in Denmark (Söderholm, 2006).

1.3.5 Estonia, Latvia and Lithuania

Detailed systems of taxes on the extraction of natural resources are in place in Estonia, Latvia and Lithuania. For a number of materials, the rates are given in Table I.1.

Table I.1 Tax rates on selected natural resources in Estonia, Latvia and Lithuania (in EUR)

Natural resource/raw material	Unit	Tax rate		
		Estonia	Latvia	Lithuania
Gravel	m ³	0.77	0.14	0.07
Sand	m ³	0.64	0.07	0.07
Limestone	m ³	1.03	0.14	0.19*
Clay	m ³	0.51	0.14	0.22
Peat	tonne	0.38	0.10-0.19	-

* per tonne.

Source: Speck *et al.*, 2006; original sources: Earth Act

(<https://www.riigiteataja.ee/ert/act.jsp?id=873617>) (Estonia); Law on Natural Resource Tax, Annex 1 (Latvia); Governmental decision on the taxes on state natural resources (Lithuania).

In Estonia, local governments are entitled to part of the revenues from the tax, their share depending on whether the mine is of local or national importance.

Between 2000 and 2004, the revenues from the Latvian natural resources tax have decreased from EUR 15 million to less than EUR 12 million, which is seen as a sign that it is on its way to achieving its goal (efficient resource use) (Speck *et al.*, 2006).

1.3.6 Italy

In Italy, a tax on sand, gravel and rock has been in operation since the early 1990s. The application of the tax is decentralised and there is no common national tax rate. Instead,

every region applies different rates at provincial and municipal levels, per cubic metre of sand, gravel and rock extracted. The revenue from the tax is accrued by the municipalities, and the legislation prescribes it should be earmarked for 'compensatory investments' in localities of quarrying activity (EEA, 2008).

1.3.7 Poland

In Poland, the extraction of mineral resources is taxed at 10% of the market price. Revenues are divided between the national, regional and local environmental funds (OECD, 2004c).

1.3.8 Sweden

The Swedish gravel tax was introduced in July 1996. Its current rate is SEK 13 (EUR 1.39) per tonne of extracted gravel. Revenues amount to EUR 22 million per year and administrative costs are some 1.7% of revenues. In contrast with the Danish raw materials tax, it does not apply to imports.

Since the introduction of the tax, there has been a strong evidence of a shift from gravel pit extraction to rock quarries (substitution of crushed rock for gravel). However, an analysis by the Swedish Geological Unit is inconclusive as to whether this shift is caused by the tax. The Ministry of the Environment was of the view that the tax had helped to sustain the shift as part of a package of policy measures. These included, among others, a tightening of the permit regime.

The overall effect of the tax had some unintended consequences. The substitution towards crushed rock required approximately three times more energy per tonne of material in comparison to natural gravel (EEA, 2008).

1.3.9 United Kingdom

The UK's aggregates levy came into effect on 1 April 2002. Its stated objectives are (1) to address the environmental costs associated with quarrying operations (noise, dust, visual intrusion, loss of amenity and damage to biodiversity) and (2) to reduce demand for aggregate and encourage the use of alternative materials where possible. There are several exemptions, among others for coal, metal ores, industrial minerals etc., for materials used in the production of lime and cement, for blocks of stone, and for aggregate necessarily arising from dredging and construction works. Exports of aggregate are also exempted, while imports are taxed upon first sale or use in the UK.

The current rate of the levy is GBP 1.95 (EUR 2.50) per tonne, amounting to some 20% of the average price of sand, rock and gravel. Administrative costs are 0.3% of the revenues. The revenues (about GBP 454 million or EUR 580 million per year) are partly earmarked for a 'Sustainability Fund'. Part of the money is used to develop quality standards for recycled aggregates. The revenue is also used for awareness-raising campaigns to encourage local authorities to purchase recycled materials when carrying out local infrastructure projects.

The United Kingdom has the highest recycling rate of aggregate materials, which accounts for almost 25 % of the UK aggregates market — the largest recycled market share of any European country (EEA, 2008).

Analysis undertaken by HM Revenue and Customs indicated a slight reduction in aggregate sales following the introduction of the aggregate tax. However, there was a lack of data over several years to show a significant and conclusive result. Industry research showed a modest shift to alternative ‘untaxed’ secondary waste materials, e.g. slate, shale and china sand. Research undertaken by the Waste Resources Action Programme (WRAP) provided evidence of an increase in recycling activity, which they predicted to continue and expand in the future. An unintended side effect was a shift of trade in aggregates, due to the proximity of Northern Ireland (with aggregate tax) to Ireland (without tax). In response, the tax rate for Northern Ireland has been reduced by 80% (EEA, 2008; UK Customs).

I.4 Taxes and charges on products

I.4.1 Belgium

Since 1996, Belgium has environmental product taxes on **batteries, disposable containers/packaging and disposable cameras**. However, these products (except non-reusable beverage containers) can be exempted from the tax if a deposit-refund scheme or a system for collection and recycling is in place that meets certain targets.

The product taxes generate revenues to the federal government (EUR 0.8 million in 2003), which are lower than the costs of collecting them. This leads the OECD (2007a) to the conclusion that, whereas Belgium has sharply increased recycling rates, these achievements seem to have come at significant cost to society.

As of 1 July 2007, a new tax was introduced on **plastic bags, plastic wrap, aluminium foil and disposable utensils**.

I.4.2 Bulgaria

According to EEA (2005b, Box 3.2) and the OECD/EEA database, Bulgaria has a product tax on **batteries**. No details are given.

I.4.3 Croatia

Croatia introduced a system of **packaging** waste charges in 2005. A distinction is made between ‘handling charges’ and ‘incentive charges’ (OECD/EEA database). Since 2006, there is also a charge on **batteries**.

I.4.4 Denmark

Denmark has had **packaging** taxes since 1979. The current system is differentiated by type of packaging, type of material, weight and volume. Since February 2004, the volume-based packaging tax on beverage containers has been differentiated to distinguish between containers for wines and spirits and those for beer and carbonated

drinks, with lower rates applying to the latter than previously. Rates for wines and spirits have remained the same since 1999, ranging according to size between DKK 0.15 (EUR 0.02) (card/laminate) and DKK 0.25 (EUR 0.03) per unit (other materials) to DKK 2.00 and 3.20 (EUR 0.27 and 0.43), respectively. The volume-based tax rates for beer and carbonated beverage containers range from DKK 0.05 (EUR 0.007) (< 10 cl) to DKK 0.64 (EUR 0.09) (> 160 cl) for all material types. Revenues from the packaging tax have increased to around DKK 900 million (EUR 120 million) in recent years. The tax is estimated to have reduced the annual waste volume by 350,000 tonnes and another 60,000 tonnes due to refilling of wine bottles (Speck *et al.*, 2006).

A separate tax on **carrier bags** came into effect in 1994. Since 2001, its rate is DKK 10 (EUR 1.34) per kg for paper bags and DKK 22 (EUR 2.95) per kg for plastic bags. Assessed by revenues collected, the tax has resulted in a reduction of consumption of materials for such bags by as much as two-thirds (Speck *et al.*, 2006).

PVC film used in food packaging is taxed at DKK 20.35 (EUR 2.73) per kg since 2001. Other products containing **PVC and phthalates** are also taxed. The tax rates are calculated on the basis of DKK 2 (EUR 0.27) per kg PVC and DKK 7 (EUR 0.94) per kg phthalate. Since 2004, rigid PVC is exempted because collection and recycling schemes operate for a range of construction products containing hard PVC. Revenues from the PVC tax varied between DKK 26 million (EUR 3.5 million) and DKK 60 million (EUR 8 million) in the years 2000-2005 (Speck *et al.*, 2006).

A tax on **disposable tableware** was introduced in 1982 and was based on value (i.e. 50 percent of the wholesale price). In 2000, the revenues from this tax amounted to DKK 65 million (EUR 9 million). Since 2001, the tax is based on weight at a rate of DKK 19.20 (EUR 2.57) per kg (Speck *et al.*, 2006).

A tax on **nickel-cadmium batteries** was introduced in April 1996. The tax applies to loose batteries and those sealed inside products. Rates have remained stable at DKK 6 (EUR 0.80) per single battery or DKK 36 (EUR 4.82) per pack for round cells joined in a unit. The purpose of the tax is twofold: to reduce the use of NiCd-batteries, and to increase recovery rates of used batteries. To pursue the latter, the revenue from the tax is used to provide the financial basis for a scheme for the collection and recovery of used NiCd-batteries (Nordic Council, 2002). Revenues from the tax fell from 31 million DKK in 1996 to 21 million DKK in 2002 in constant prices, leading the Ministry of Finance to conclude that the tax has been effective. (Speck *et al.*, 2006).

1.4.5 Estonia

A tax on **packaging** is applied in Estonia, with rates depending on the material (EUR 0.64 per kg for glass and ceramics; EUR 2.56 per kg for plastics and metals; EUR 1.28 for paper and other materials). Exemptions are possible if certain recovery percentages are achieved (Speck *et al.*, 2006).

1.4.6 Finland

A tax on **non-refillable beverage containers** has been in place since 1993. It complements the DRS for refillable and non-refillable beverage containers (described in Section 5.6). The tax rate is EUR 0.51 per litre. Beverage packagings which are recycled

and enter a functional return system approved by the government are not subject to the tax. Revenues from the tax were budgeted at EUR 13 million for 2005 (Speck *et al.*, 2006). According to Hiltunen (2004), the tax has resulted in a near complete recycling rate of packaging for soft drinks. Research conducted by the Pirkanmaa Regional Environment Institute showed that in 2006 almost 98% of all of the refillable drinks packages used in Finland were returned, as well as 88% of packagings whose materials can be recycled (Finland's environmental administration).

1.4.7 France

In September 2008, the French government announced the introduction of new environmental taxes on various products, including a tax on **disposable tableware** of EUR 0.90 per kg. However, following public protest the plan for this 'picnic tax' was rapidly withdrawn (Reuters).

1.4.8 Hungary

Hungary has a system of environmental product taxes and charges which covers, among others, **packaging materials** and **batteries**, **paper materials for advertising**, **refrigerators** and **electronic devices** (EIONET; OECD, 2008).

1.4.9 Iceland

Iceland applies a system of 'recycling fees' on various products. These fees feed into a recycling fund in order to finance the collection, transport, recycling, recovery or disposal of the waste from the products on which they are levied. Most recycling fees apply to products creating hazardous waste (e.g. **batteries**), but recently the scheme has been expanded to cover **packaging** as well (Speck *et al.*, 2006).

1.4.10 Ireland

The Irish tax on **plastic bags**, introduced in 2002, seems to be one of the biggest 'success stories' in the area of waste related product taxes. It is even referred to (although with a question mark) as 'the most popular tax in Europe' (Convery *et al.*, 2007). The initial tax rate was EUR 0.15 per bag; in 2007 it was increased to EUR 0.22. The tax has led to a reduction in the consumption of plastic shopping bags in excess of 90%. Administrative costs are low (some 3% of revenues). The acceptance of the tax seems to have been enhanced by the fact that the revenues are ring-fenced into an Environmental Fund (McDonnell *et al.*, 2008).

1.4.11 Italy

According to EEA (2005b, Box 3.2), Italy has product taxes on **batteries** and on **disposable containers**. According to the OECD/EEA database, there is also a tax on **plastic bags** that are not biologically decomposable.

1.4.12 Latvia

Latvia has a tax on **packaging** and **disposable tableware**. Table I.2 presents the tax rates for 2000-2005. Tax exemptions apply if recovery or recycling of packaging waste is undertaken and if companies implement voluntary programmes for packaging waste management.

Table I.2 Tax rates on packaging material and disposable tableware in Latvia

Material	Tax rate (EUR/tonne)
Paper/cardboard and wood or other materials of natural fibre	17
Paper/cardboard and other laminated materials with polymer and metal components	86
Plastic, except PET	129
PET	150
Metal	86
Glass	57

Source: Speck *et al.*, 2006; original source: Law on Natural Resource Tax, Annex 8.

Charges apply to a number of other products, including **batteries** (15% of the value) and **fluorescent lamps** containing mercury (LVL 0.14 or EUR 0.20 per item). The main purpose of these product charges is to stimulate the collection and recycling of certain problematic waste streams financially, as well as to encourage a reduction in the consumption of these products by increasing their sales price (Speck *et al.*, 2006).

1.4.13 Lithuania

Lithuania introduced a tax on **packaging** materials in 2003. The rates are given in Table I.3. A (partial) exemption from the tax is granted if certain reuse and recycling targets are achieved.

Table I.3 Tax rates on packaging material in Lithuania

Material	Tax rate (EUR/tonne)
Paper and cardboard	29
Plastic	521
Metal	753
Glass	26
Combined	579
Other	58

Source: Speck *et al.*, 2006; original source: Law on the Environmental Pollution Charge.

Charges apply to a number of other products, including **batteries** (4% of the wholesale price) and **fluorescent lamps** containing mercury (15% of the wholesale price). The main purpose of these product charges is to stimulate the collection and recycling of certain problematic waste streams financially, as well as to encourage a reduction in the consumption of these products by increasing their sales price (Speck *et al.*, 2006).

I.4.14 Norway

An environmental tax on **beverage containers** was introduced in Norway in 1974. In 2000 it was differentiated according to the type of material the container is made of. In 2005, the rate per unit for glass and metal containers was NOK 4.46 (EUR 0.55), for plastic NOK 2.69 (EUR 0.33) and for cartons NOK 1.11 (EUR 0.14).

The tax is reduced according to the percentage of containers that are recycled, up to 95 percent, at which level the container-type is exempt from the tax. Exemptions exist on juice and milk products and other products typically consumed in the home. Additionally, since 1994, a flat rate tax has applied to **non-refillable beverage containers**. The rate of this tax amounted to NOK 0.89 (EUR 0.11) per unit in 2005 (Speck *et al.*, 2006). This tax is independent of recycling rates. Containers for milk and milk products and beverages made of cocoa and chocolate and concentrates of these are exempted from this tax (Nordic Council, 2002).

I.4.15 Poland

A number of environmental product charges exist in Poland, among others on **packaging, batteries, discharge lamps and electr(on)ic equipment** (OECD/EEA database).

I.4.16 Portugal

In Portugal, charges on **batteries and packaging** are applied from which the collection and treatment are financed (OECD/EEA database).

I.4.17 Romania

On 1 January 2009, a tax on non-biodegradable **plastic bags** will be introduced in Romania. The rate will be RON 0.2 (EUR 0.05) per bag. The tax should be clearly visible on the sales slip. Revenues go to the Environmental Fund (EVD).

I.4.18 Slovakia

A system of taxes (charges) exists from which a dedicated 'Recycling Fund' is financed. Table I.4 shows the tax rates on various products.

Table I.4 Recycling Fund charge rates in Slovakia

Product/Material	Charge rate (EUR/tonne)
Batteries	200 - 5000
Consumer electronics	300 - 1320
Lamps with mercury	370 – 390
Paper and cardboard	160
Iron and steel packaging	30
Aluminium packaging (and alloys)	110
Glass	16
Multilayer combined materials	2030

Source: EIONET

1.4.19 Sweden

Packaging charges are levied in Sweden to finance the system of producer responsibility. The rates are as in Table I.5.

Table I.5 Charge rates on packaging material in Sweden

Material	Charge rate (EUR/tonne)
Paper and cardboard	450
Corrugated cardboard	20 – 750
Plastic bags	1500
Other plastic (incl. foamed)	2000
Metal (aluminium and steel-plate)	600 - 1200
Metal (drums)	60

Source: Speck *et al.*, 2006

A study published in 2005 revealed that packaging and waste paper, which should come under the responsibility of producers, comprises approximately 30 percent of household waste. Producer take-back was found to lie at 55 percent of the packaging and waste paper that enters the market (Speck *et al.*, 2006).

Environmental charges levied on producers and importers of **batteries** should cover the costs to the municipalities of the collection scheme, for instance, the disposal, treatment or recovery activities associated with environmentally hazardous batteries as well as information dissemination with regard to operation and promotion of the scheme. The rate for cadmium batteries is EUR 33 per kg; for alkaline, silver oxide and zinc-air batteries EUR 55 per kg (Speck *et al.*, 2006).

1.4.20 Switzerland

Switzerland applies ‘prepaid disposal fees’ to a number of products, including **packaging**, **batteries** and **electr(on)ic appliances** (OECD/EEA database).

1.4.21 Canada

The provinces of Manitoba and Ontario have taxes on **non-refillable containers** (OECD, 2004b).

1.4.22 China

In 2006, China has introduced a 5% consumption tax on **disposable wooden chopsticks**, of which China uses ten billion boxes each year, and exports another six billion, requiring 1.3 million cubic metres of timber from the country’s forests. A 5% tax on **wooden floor panels** will also discourage the consumption of timber resources (OECD, 2007b).

Since 1 June 2008, retailers in China are obliged to charge their customers a fee for **plastic bags** that they provide them with. The fee should at least be equal to the cost price of the bag. This measure is introduced for the purpose of saving resources, protecting the ecological environment and guiding consumers to reduce the use of plastic

bags (Lawinfochina). Although strictly speaking it is not a tax or charge, the measure has a comparable function in that it is intended to reduce demand for a waste generating product by increasing its price above the 'free market' level.

1.4.23 South Korea

Since 1993, South Korea applies a number of product charges on 'hard to recycle' products, including **chewing gum**, **cosmetics containers** (glass bottles), and **diapers**. Since 2003, the charge scheme also applies to **plastic** used for packaging, construction, furniture and toys. The rates have been too low to create incentives to move towards recyclable products. Instead, the charges serve to raise revenues that accrue to the budget of the Ministry of Environment (OECD, 2006c).

1.4.24 United States

Some states (e.g. Nebraska, New York and Virginia) apply taxes on the sales of **litter-generating products** (OECD/EEA database).

1.5 Deposit-refund schemes

1.5.1 Introduction

This section only covers mandatory deposit-refund schemes with a legal basis; voluntary schemes (which are common in most countries) are not discussed. It should be noted, however, that in some countries (such as Norway) the use of (voluntary) deposit-refund schemes is strongly stimulated by the fact that high return rates allow companies to enjoy a reduced rate or exemption from packaging taxes.

1.5.2 Austria

Austria has a mandatory deposit-refund system for refillable plastic beverage bottles (EEA, 2005a).

1.5.3 Czech Republic

The Act on Packaging lays down a detailed system of returnable packaging for which deposits are made. Government Regulation 111/2002 Coll., laying down the amount of the deposit for selected types of returnable packages for which a deposit is made, comprises a list of returnable packages for which deposits are paid, together with the amount of such deposits. The list contains seven types of bottles (source: EIONET).

1.5.4 Denmark

Denmark has a mandatory deposit-refund system for beverage containers. Presently it covers beer and soft drinks, but it will be extended to mineral water (Miljøministeriet, 2008). Since 2002 (when the ban on drinks in cans was lifted) it also applies to non-refillable containers. Deposit rates vary between DKK 1 (EUR 0.13) and DKK 3

(EUR 0.40), depending on size and material (glass, plastic, can). In 2003, return percentages were 99 percent and 80 percent for refillable and non-refillable beverage containers, respectively (Speck *et al.*, 2006). By 2005, the latter figure had increased to 84 percent (EIONET).

1.5.5 Estonia

Since 1 May 2005, a mandatory deposit-refund system for beverage containers is in force. Deposit rates are set by the Ministry of Environment. The rates amount to EUR 0.032 or 0.064, depending on material, volume and (non-)reusability (Speck *et al.*, 2006).

1.5.6 Finland

A mandatory deposit-refund scheme applies to both refillable and non-refillable beverage containers for reuse and recycling, respectively. Deposit rates are presented in Table I.6.

Table I.6 Deposit rates on beverage containers in Finland

Containers	EUR per unit
Glass (0.33, 0.35 and 0.5 litre)	0.10
Glass (1.0 litre)	0.40
PET (0.5 litre)	0.20
PET (1.0 and 1.5 litre)	0.40
Crates (24 x 0.33 litre)	2.20
Aluminium cans	0.15

Source: Speck *et al.* (2006); original sources: www.palpa.fi and www.ekopullo.fi.

The current rate of return of glass bottles for beer and soft drinks has been at between 98 and 99 percent for a number of years. However, the collection rate for beverage cans with deposit was only 82 percent in 2004 (Speck *et al.*, 2006).

1.5.7 Germany

In 2003, a mandatory deposit of EUR 0.25 was introduced for non-refillable packaging of beer, soft drinks and water. As a result, after a steady decline of reusable beverage packaging put on the market until 2002, there was an substantial increase in 2003 (EIONET).

1.5.8 Iceland

A deposit-refund system on refillable and non-refillable aluminium, steel, plastic and glass packaging has been in place in Iceland since 1989. Deposit rates were between ISK 7.23 (EUR 0.07) and ISK 10.63 (EUR 0.10) during the period 2002-2005, depending on material and size. The system has achieved a return rate of over 80 percent for refillable and nonrefillable beverage containers, including cans. Since the introduction on 1 January 2008 of recycling fees (see Section 4.9) on non-refillable aluminium, steel, plastic and glass packaging, the deposit-refund system no longer applies to these types of containers (Speck *et al.*, 2006).

1.5.9 Lithuania

A governmental decision on deposit-refund systems came into force in 2003. A deposit must be collected for 0.5 l and larger re-usable glass packaging for beverages (beer, alcohol, soft drinks, mineral water and juice) (Speck *et al.*, 2006).

1.5.10 Sweden

Sweden has a long-standing system of deposit-refund schemes for both refillable and non-refillable beverage packagings. Table I.7 shows the deposit rates and collection rates.

Table I.7 Deposit rates and collection rates for beverage containers in Sweden

Containers	Deposit rate (EUR per unit, 2005)	Collection rate (2002)
Glass (0.33 litre, refillable)	0.07	99%
Glass (0.50 litre, refillable)	0.09	
Large PET (refillable)	0.45	
Crates for 33 cl	2.50	97%
Crates for 50 cl	3.10	
Aluminium cans (for recycling)	0.06	
PET – small (for recycling)	0.10	78%
PET – medium (for recycling)	0.20	

Source: Speck *et al.* (2006).

1.5.11 Canada

Several Canadian states run beverage container recycling programs, which include deposit-refund schemes (OECD/EEA database). The province of Yukon's deposit-refund scheme, in existence since 1992, covers a large range of beverage containers (including aluminium cans), with refund rates for some types lower than the deposits (presumably to cover handling costs). Return rates of nearly 85% were reported for 2000-2001 (EPA, 2004).

1.5.12 South Korea

South Korea has deposit-refund schemes in place for various products, including medicine and detergent packaging, batteries and home appliances. However, unlike deposit-refund systems in the United States and Europe, producers and importers, not consumers, pay the deposits into a 'Special Account for Environment Improvement'. They are required to collect and treat their waste and the Account reimburses them according to the recovery rate achieved (ILSR).

1.5.13 United States

In the US, ten states use mandatory deposit-refund systems on beverage containers to encourage recycling (EPA, 2001; OECD, 2006e). Although data are incomplete, anecdotal evidence suggests that beverage container deposit laws have significantly reduced litter in several states. For example, Oregon reported a 75% to 85% decrease in

roadside litter just two years after enacting deposit legislation. Another probable impact has been an increase in the percentage of containers recycled, although this is difficult to confirm due to a lack of historical data on recycling. There is also a relatively high market share for refillable containers in states with deposit schemes. On the other hand, EPA (2001) notes that the costs of deposit systems may be substantial and could also divert revenues from, and lower the cost effectiveness of, curbside recycling programs.

Some states have (had) ‘advanced disposal fees’ on products such as containers (Florida) and ‘white goods’ (North Carolina) (EPA, 2001).

1.6 Subsidies and fiscal incentives

1.6.1 Belgium

Flemish municipalities are encouraged to contribute to the national waste policy objectives by means of subsidies that are significant for ‘early adopters’ and decrease in the course of time (Gellynck and Verhelst, 2007).

Household composting is stimulated in Flanders by various means, including subsidies for composting vessels. By 2006, 38% of Flemish households were engaged in composting at home (MIRA).

In Belgium, a reduced VAT rate is applied to supplies of certain recovered materials and by-products (EC, 2008). The application of this reduced VAT rate is restricted to so-called ‘recycle shops’, which provide employment to low-skilled unemployed people.⁵⁷

1.6.2 Czech Republic

In the Czech Republic, properties and structures serving exclusively for operation of recycling activities are exempted from the real estate tax (MoE, 2004, cited by UCD).

When VAT was introduced in the Czech Republic in 1993, a reduced rate was applied to certain environmentally preferable product categories, including paper and cellulose products made with minimum 70% recycled paper. This scheme had to be abolished with the Czech Republic’s EU accession in 2004.

1.6.3 Denmark

The Danish Environmental Protection Agency administers a subsidy scheme associated with the collection of nickel-cadmium batteries for recycling. The rate is DKK120 (EUR 16) per kg (excl. purchase tax) to professional collectors (Speck *et al.*, 2006).

1.6.4 France

In France, waste collection services are normally taxed at the standard VAT rate of 19.6%. However, the reduced rate of 5.5% is applied to the collection of separate waste

⁵⁷ This is made possible by Annex III of the VAT Directive (2006/112/EC) which allows (under item 15) the application of the low VAT rate to the “supply of goods and services by organisations recognised as being devoted to social wellbeing by Member States and engaged in welfare or social security work”.

streams, providing an incentive to local authorities to put in place separate collection schemes (EIONET).

1.6.5 Poland

Until the end of 2006, Poland has applied a temporary excise duty exemption for liquid fuels produced from plastic waste (ENDS, 2006).

1.6.6 Sweden

To promote the sorting of waste in multi-storey houses, the house-owner could apply for an investment subsidy during 2005 and 2006 (EIONET).

1.6.7 United Kingdom

A Waste Minimisation and Recycling Fund was set up in 2001 to provide assistance to local authorities to improve waste management and increase recycling (EEA, 2005a).

The West Sussex 'real nappy scheme' provides free laundry services for families using the scheme. The results of the scheme show that 500 families participated in the scheme between 1999 and 2000 and this saved the production of 800 tonnes of disposable nappies as well as cost savings of EUR 32,000 for the local authority. The waste reduction per family that used the scheme translated to 1600 kg per family per year and the incentive provided amounted to EUR 48 per family per year (Husaini *et al.*, 2007)..

1.6.8 EU

Article 106 of the VAT Directive (2006/112/EC) allows Member States to apply a reduced VAT rate to certain labour intensive services, until the end of 2010.⁵⁸ These include small services of repairing bicycles, shoes and leather goods, and clothing and household linen. Although the primary objective of this scheme is to promote employment and discourage 'black market' activities, it can also be seen as a fiscal incentive for waste prevention and recycling. Currently, the Benelux countries, Ireland and Poland are the only Member States using this option for the mentioned services.⁵⁹

1.6.9 Australia

Australia exempts certain recycled paper products from its Wholesale Tax (EPA, 2004). In most Australian States, some of the revenue generated from landfill levies is hypothecated to waste management programs, including recycling projects (MMA and BDA Group, 2003).

⁵⁸ On 7 July 2008, the Commission has presented proposals to give these options a permanent basis.

⁵⁹ In addition, Greece applies a reduced rate only to clothing and household linen repair. Source: EC (2008).

1.6.10 Japan

Japan has a scheme of accelerated depreciation for investments in recycling equipment (OECD/EEA database).

1.6.11 South Korea

South Korea has a system of soft loans for fostering the recycling industry (OECD/EEA database).

1.6.12 United States

At least 24 states in the US have grant or loan programs that promote the recycling industry. Twenty-eight states have offered tax incentives for businesses that recycle used products. Many states also have a policy of preferential procurement of recycled products. Under such policies, states are prepared to pay a specified percentage (usually 5 or 10%) more for products with recycled content than for comparable products that do not contain recycled materials (EPA, 2001).

Several municipalities in the eastern United States participate in the 'RecycleBank' programme. This private enterprise collects recyclable (but otherwise unseparated) waste from households and rewards them with coupons (worth up to USD 400 per year) that can be spent at certain national retail chains. The system is financed by the participating municipalities, which get a guarantee that they will save at least that much in disposal fees as waste is diverted from landfills and incinerators (DeSimone, 2006).

1.7 Trading schemes

1.7.1 United Kingdom: Packaging Recovery Notes

In the wake of the EU packaging and packaging waste directive, the UK government tasked industry with devising mechanisms through which packaging recycling and recovery targets would be met. When it became clear that this process was too fractious to lead to a clear outcome, the government stepped in and implemented a system in which the companies obligated under the relevant legislation would have to provide evidence to authorities that they had recycled and recovered the required amount of packaging waste. The form of evidence is known as the packaging recovery note (PRN).

There is some dispute about whether the PRN system was designed to be a tradable credit system, but in practice a *de facto* trading system has emerged. The degree to which the UK system can be considered successful is a matter of some debate. Nevertheless, EEA (2005b) concludes that the system does, in essence, work.

1.7.2 United Kingdom: landfill allowance trading

The landfill allowance schemes have been designed to enable the UK to meet the Landfill Directive targets for the reduction of biodegradable municipal waste (BMW) sent to landfill. Allowances are allocated to waste disposal authorities (WDAs) and unitary authorities (UAs). The total number of allowances decreases gradually until

2020. Each administration has developed its own landfill allowance scheme, to take account of local circumstances and priorities. Trading of allowances is only permitted in England (since 2005) and Scotland (since 2008). In England, allowances can be traded between WDAs and UAs. In addition, 'banking' and 'borrowing' of allowances between years is possible, within certain limits. There is a penalty of GBP 150 (EUR 192) per tonne if a WDA breaches its landfill allowances target in the scheme year. In 2006-2007, 18 trades took place in England, at an average allowance price of GBP 17.67 (EUR 22.62) per tonne (Environment Agency, 2007).

Calculations by Barrow (2003) suggested that in favourable circumstances, the trading scheme could reduce compliance costs by over 50 per cent in the initial years, declining over time. However, there were several uncertainties associated with these estimates.

Appendix II. Trading schemes

II.1 Introduction

The use of tradable permits in environmental policy has its origins in the United States, but has become important in Europe as well with the introduction of greenhouse gas emissions trading in 2005. In the area of waste policy, however, the application of this instrument is still very limited.

II.2 Theory and literature

As with other policy instruments, the impact of a tradable permit scheme will to a large extent depend on the specific design. Crucial design parameters in trading schemes include, among others:

- The choice between a ‘cap-and-trade’ system (i.e., there is an absolute amount of emissions waste that can be produced, landfilled etc.) or a ‘performance standard based’ system (i.e., allowances can be sold if a firm performs better than a certain predetermined standard);
- The choice between ‘grandfathering’ and auctioning as the mechanism for initial allocation;
- Any restrictions on trading (e.g. concerning the actors that are allowed to trade, trading between different regions and years, validity of allowances upon plant closure, etc.).

The experiences that have been gained with various kinds of emissions trading (see e.g. Tietenberg, 2006) may contain useful lessons for possible applications in the area of waste management.

II.3 The Dutch situation

In the Netherlands, tradable permits are not being used as a policy instrument for waste management. During the preparation of the National Waste Management Plan (LAP) 2002-2012 (VROM, 2007) the possible use of tradable permits to reduce the landfilling of combustible waste has been studied. It was concluded that the introduction of this instrument would not be possible at short notice and that it would not have any added value compared to the instruments that were already in place.

II.4 Foreign experiences⁶⁰

The use of tradable permits in waste policy is largely confined to the UK, where two schemes exist:

- A ‘Packaging Waste Recovery Notes’ (PRN) system, which allows companies to comply with their recovery and recycling obligations by buying PRNs from accredited reprocessors that actually carry out the recovery and recycling;

⁶⁰ For details, see Appendix I.

- Landfill allowance trading, creating the opportunity for waste disposal authorities to exceed their allocated amount of biodegradable municipal waste going to landfill if they buy additional allowances from other authorities that do not need their full quota.

II.5 Synthesis

Tradable permits are as yet hardly used in waste policy. Their main role could be to ensure an efficient allocation of a certain fixed quatum, e.g. landfill or incineration capacity. However, they are less suited as an instrument to promote waste minimisation and will therefore not be analysed further in this study.

Appendix III. Exchange rates used

Denmark: DKK 1.00 = EUR 0.134

Iceland: ISK 1.00 = EUR 0.009

Latvia: LVL 1.00 = EUR 1.44

Norway: NOK 1.00 = EUR 0.126

Poland: PLN 1.00 = EUR 0.29

Sweden: SEK 1.00 = EUR 0.107

United Kingdom: GBP 1.00 = EUR 1.28

United States: USD 1.00 = EUR 0.65